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IDEAS AND PERSPECTIVES

Mixed biodiversity benefits of agri-environment schemes in five European countries

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Abstract

Agri-environment schemes are an increasingly important tool for the maintenance and restoration of farmland biodiversity in Europe but their ecological effects are poorly known. Scheme design is partly based on non-ecological considerations and poses important restrictions on evaluation studies. We describe a robust approach to evaluate agri-environment schemes and use it to evaluate the biodiversity effects of agri-environment schemes in five European countries. We compared species density of vascular plants, birds, bees, grasshoppers and crickets, and spiders on 202 paired fields, one with an agri-environment scheme, the other conventionally managed. In all countries, agri-environment schemes had marginal to moderately positive effects on biodiversity. However, uncommon species benefited in only two of five countries and species listed in Red Data Books rarely benefited from agri-environment schemes. Scheme objectives may need to differentiate between biodiversity of common species that can be enhanced with relatively simple modifications in farming practices and diversity or abundance of endangered species which require more elaborate conservation measures.

Keywords

Biodiversity conservation, European Union, farmland wildlife, land-use intensity, policy evaluation, Red Data Book species.

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INTRODUCTION

In many parts of Europe, agricultural landscapes are well over 2000 years old (Williamson 1986; Groppali 1993). Over time, many species of wildlife have adapted to these extensively managed and highly variable landscapes resulting in the development of many anthropogenic species-rich ecosystems. Simultaneously, continued human population growth and the associated need for land caused the destruction of most European undisturbed natural ecosystems. For some species this resulted in the loss of their primary habitat, making them almost completely dependent on their secondary, agricultural habitats for continued survival (e.g. Great Bustard *Otis tarda*, Grey Partridge *Perdix perdix*, Black-tailed Godwit *Limosa limosa*). Over the last few decades, agricultural changes aimed at making farming more cost-effective have had accelerating adverse effects on wildlife (Donald *et al.* 2001; Benton *et al.* 2002). This not only caused a reduced biodiversity of most agro-ecosystems, it also resulted in many European farmland-inhabiting species becoming threatened (Tucker & Heath 1994).

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Agri-environment schemes aim to counteract the negative effects of modern agriculture on the environment by providing financial incentives to farmers for adopting environmentally friendly agricultural practices. Agri-environment schemes are considered the most important policy instruments to protect biodiversity in agricultural landscapes (EEA 2004). Additionally, agri-environmental subsidies do not distort international trade and give European countries the possibility to continue supporting the farming community when direct agricultural subsidies are under pressure (Swinbank 1999; EU 2005). All member-countries of the European Union (EU) are currently obliged to develop and implement agri-environment programs. Investments in agri-environment schemes are substantial. Schemes currently cover *c.* 25% of all farmland in the 15 older EU countries (EU 2005) and in 2003, the estimated agri-environmental budgets of EU-member states alone amounted to €3.7 billion (EEA 2002). Not all schemes are aimed at biodiversity conservation, but this is one of multiple objectives of many schemes and a significant number of European agri-environment schemes specifically addresses this objective (Kleijn & Sutherland 2003).

Currently, there is a lively debate on whether agri-environment schemes are effective in producing ecological benefits on farmland (Aebischer *et al.* 2000; Kleijn *et al.* 2001; Peach *et al.* 2001; Kleijn & Sutherland 2003; Bradbury *et al.* 2004; Vickery *et al.* 2004; Tscharntke *et al.* 2005). Studies that examine the ecological effects of agri-environment schemes are still few, especially when considering the large sums of money that are spent on schemes annually (Kleijn & Sutherland 2003), and the lay-out of many of these studies is unsuitable for drawing reliable conclusions. Lack of knowledge of the responses of farmland wildlife to the implementation of agri-environment schemes makes it impossible to perform cost–benefit analyses or to improve scheme performance.

The ecological effects of schemes are strongly influenced by the way in which a scheme is designed and implemented. Scheme design is not only determined by the prescriptions that follow from the ecological requirements of target species groups, but is usually the outcome of a process of careful balancing of a range of ecological, socio-economic, administrative and political interests (Buller *et al.* 2000). The outcome is usually not optimal from a conservation perspective. For example, participation to agri-environment schemes is on a voluntary basis and, in most countries, the basic units for participation are individual fields. This often results in an erratic spatial distribution of fields with agri-environment schemes in an otherwise intensively farmed landscape. This may reduce the effectiveness of the measures, because populations may not be able to disperse from one field to the next (Geertsema 2005). Additionally, participating farmers are committed for periods of only 5 or 6 years (occasionally 10 years), after which they are free to stop. On intensively used farmland, the restoration of species-rich communities following the re-instatement of more extensive management may take considerably longer than this (Olff & Bakker 1991; Walker *et al.* 2004) so that farmers may need to participate for several contract periods before significant effects are visible. The success of schemes therefore depends to a large extent on the continued motivation of farmers to participate.

Examining the *in situ* ecological effects of agri-environment schemes can most effectively be performed by comparing trends in biodiversity on treatment fields and control fields both before and after implementation of the treatment (a replicated before–after control-impact or BACI approach, e.g. Bro *et al.* 2004). However, a number of complications exist that are characteristic to studies aiming to determine the ecological effects of farm management in the field (e.g. effects of genetically modified crops or integrated pest management). Baseline data are rarely available and usually an estimate has to be made of management that has been in place for quite some time already. Farmers usually participate in schemes preferentially with fields less suitable for intensive farming (Kleijn *et al.* 2004) and these may support higher biodiversity than the surrounding fields (Kleijn & van Zuijlen 2004). Comparing biodiversity on sites with and without agri-environment schemes at one point in time does not distinguish between possible initial differences and differences attributable to the schemes and results may therefore be biased (Kleijn & Sutherland 2003). Comparing time trends after the start of conservation management partially solves this problem. However, over time management on the target sites often changes making an increasing number of study sites unsuitable for the purpose of the evaluation. Furthermore, the ecological objectives of most agri-environment schemes are vaguely formulated (Schramek 2001; Table 1). It is therefore unclear how biodiversity should be estimated and the outcome of evaluation studies depends to a large extent on what species groups are being surveyed and what index of diversity is being used.

We present a study approach to evaluate the biodiversity effects of conservation management on farms that can be applied uniformly and effectively in a wide range of agricultural landscapes. This approach has been implemented in five European countries to examine the biodiversity effects of agri-environment schemes. Biodiversity was estimated as the species density and abundance of vascular plants, bees (Hymenoptera: Apoidae), grasshoppers and crickets (Orthoptera), spiders (Aranae) and birds (Aves). These five species groups occupy different trophic levels and consist of a wide range of species-rich taxa. To further evaluate the conservation value of these agri-environment schemes, we distinguished between the response of: (i) all

Table 1 Summary of the main biodiversity related objectives and prescriptions of the agri-environment schemes that were ecologically evaluated in three areas in each country

Germany

Evaluated scheme: organic farming in arable cropping systems

Scheme objectives: to care for the natural basics of the life of the soil, water and air. To actively protect nature and preserve species. To prevent damage to the environment

Scheme prescriptions: no synthetic fertilizers and pesticides or organic fertilizers from conventional farms. No use of genetically modified organisms. No post-harvest treatments of seeds and plant materials with synthetic disinfectants. Maintenance and creation of habitats for beneficial organisms is stimulated

Study areas: Leine Bergland (Bundesland Niedersachsen), Lahn-Dill Bergland (Hessen) and Soester Boerde (Nordrhein-Westfalen)

Mean age (SE) of schemes on study sites: 12.4 (1.3) years

Spain

Evaluated scheme: measures to protect steppe-living birds in extensively managed cereal fields and compensation measures in the buffer area of the Cabañeros National Park

Scheme objectives: the conservation of steppe-associated birds in Spain

Scheme prescriptions: maximum annual fertilizer applications 60 kg N ha^{-1} and maximum pesticide applications 1.5 kg ha^{-1} of AAA-type pesticides. No agricultural activities between 1 April and 31 May, 15 April to 25 June or 15 April to 10 July depending on area. No ploughing of stubble until 1 August or fallow until 15 October if the fields are to be cultivated in the following year. Keeping stubble until 1 February or 1 March, depending on area, if fallow is the next stage of a field's cropping rotation. No use of dressed seeds and no burning of fallow vegetation. Strips covering 3% of the fields are left unploughed, except in the buffer area of Cabañeros

Study areas: Retuerta del Bullaque (Ciudad Real Province, Castilla La-Mancha Region), Huecas (Toledo Province, Castilla-La Mancha) and La Guardia (Toledo, Castilla-La Mancha)

Mean age (SE) of schemes on study sites: 6.7 (0.1) years

Switzerland

Evaluated scheme: Ecological Compensation Areas aimed at the conservation of extensively used hay meadows

Scheme objectives: natural biodiversity should be enhanced. Agro-biodiversity should be preserved. The decline of endangered farmland species should be stopped or reversed by 2005, compared with reference period 1990–1992. Note: no biodiversity data are available for the years 1990–1992

Scheme prescriptions: no fertilizer applications. No pesticide applications other than patch-wise control of problem-weeds. Vegetation cut and removed at least once a year but not before 15 June (lowlands, foothills of Alps) or 1–15 July (mountains, date depending on region)

Study areas: Ruswil (Canton of Luzern), Flühli (Luzern) and Bauma (Zürich)

Mean age (SE) of schemes on study sites: 6.6 (0.6) years

The Netherlands

Evaluated scheme: meadow bird agreements in wet grasslands

Scheme objectives: the conservation of Dutch meadow birds

Scheme prescriptions: no agricultural activities between 1 April and 1–15 June (depending on tier). No changes in field drainage and no pesticide applications other than patch-wise control of problem weeds. Additionally, on surrounding fields farmers face no restrictions but are paid for each meadow bird clutch laid on their land (per-clutch payment scheme; Musters *et al.* 2001). These fields were only surveyed for birds

Study areas: Eempolder (Province of Utrecht), Utrechtse en Hollandse Venen (Utrecht, Zuid-Holland), Alblasserwaard and Vijfheeren-landen (Zuid-Holland)

Mean age (SE) of schemes on study sites: 5.7 (0.4) years

UK

Evaluated scheme: 6-m-wide grass field margin strips along arable fields; one of a number of schemes available in the Countryside Stewardship Scheme

Scheme objectives: create networks of uncropped grass margins and areas of wildlife seed mixtures, to provide wildlife habitats and corridors to buffer habitats and features from agricultural operations

Scheme prescriptions: create field margin strips through natural regeneration or sowing of grass or grass/forbs mixture. Vegetation mown once a year after mid-July; dense cuttings must be removed. No pesticide applications other than patch-wise control of problem weeds. Margins may not be used for regular access by farm vehicles

Study areas: field pairs in the UK were located in open, closed and intermediate landscapes scattered over the counties Avon, Dorset, Gloucestershire, Somerset and Wiltshire

Note: as the 6-m-wide grass margin strip scheme does not influence farming practices in the cropped field, edge samples in the UK were taken in the pre-existing boundary to examine whether increased area of semi-natural habitat and buffering against agrochemicals results in enhanced biodiversity

Mean age (SE) of schemes on study sites: 3.5 (0.1) years species; (ii) uncommon plant and arthropod species; and (iii) Red Data Book species. The approach was used to investigate in all five countries, first, whether species density of the examined species groups was higher on fields with agri-environment schemes compared with conventionally managed fields and, second, whether species density or abundance of uncommon or endangered farmland species was enhanced on fields with agri-environment schemes. The ecological implications and conservation perspectives of the results are discussed and potentials as well as limitations of the study approach are highlighted.

METHODS

General approach

Effects of agri-environment schemes can most reliably be determined if (trends in) species density on scheme sites are compared with control sites that are similar to scheme sites in every respect except management. As environmental heterogeneity increases with the size of the area, evaluation studies performed at higher spatial scales make it increasingly difficult to separate the effects of scheme implementation from effects of confounding environmental factors. Furthermore, agri-environment schemes are being implemented at the scale of agricultural fields. We therefore compared biodiversity on pairs of similar-sized fields, one field with an agri-environment scheme, the other conventionally managed. Prior to pairing the fields, maps indicating soil type and groundwater level were carefully inspected and areas were visited to examine landscape context of potential study sites. Only fields that had similar environmental conditions, other than those influenced by the schemes were paired.

The studied scheme sites should be a random sample of all scheme sites, so that the results are representative for the scheme in general. This furthermore ensures that not only the effectiveness of agri-environmental measures is tested, but also the way in which a scheme is being implemented (e.g. zonal vs. horizontal approach, see Kleijn & Sutherland 2003). We selected three independent areas in each country and in each area seven field pairs were located. Selection of pairs was random as far as the availability of scheme sites and comparable control sites allowed.

Evaluated agri-environment schemes should be in place as long as possible to allow effects of management prescriptions to develop. However, schemes need not necessarily be older than the contract period, i.e. 5 years, as effects that have not (yet) become apparent in this period are of questionable use to agri-environment schemes as a nature conservation tool. The mean age of agri-environment schemes on the examined sites was in the range of 3.5–12.4 years in the various countries (Table 1).

Agri-environment programmes differ between countries and there are no schemes that are implemented in different countries in exactly the same way. Therefore, we studied the effects of a different, widely implemented scheme on biodiversity in each country (scheme details are given in Table 1). Because management prescriptions differ between countries no straightforward comparisons can be made between the effects of schemes in different countries. However, the exploration of qualitative effects between countries may reveal general patterns in factors affecting scheme success.

Sampling protocol

In 2003, cover of individual plant species and total number of plant species were determined in 20 plots of 5 × 1 m on each field. Ten plots, spaced 5 m apart, were located in and parallel to the field edge, 10 more were located similarly in the field center. Bee species density and abundance was estimated in three survey rounds in late spring and summer. In each round, bees were caught in edge and centre using sweepnet surveys (60 sweeps per location per round) and 1-m wide transect surveys (Banaszak 1980; 15 min per location per round). Orthoptera (grasshoppers and crickets) were surveyed once in late summer, when adults were present, using the same methods as for bees. Bee and Orthoptera sampling was performed between 10.00 and 16.00 hours on sunny days. Spiders were surveyed using one pitfall trap in the edge and one in the centre of the field. To ensure that traps in different countries were opened during approximately the same phenological period, traps were opened 2 weeks after full bloom of *Taraxacum officinale* GH Weber ex Wiggers (dandelion). Trapping was performed in two consecutive 2-week periods followed by a final 2-week period separated by a 2-week interval in which traps were closed (6 weeks in total; Duelli *et al.* 1999). In agricultural landscapes, this approach yields *c.* 70% of the species caught after a full seasons sampling (Duelli *et al.* 1999). All arthropods were caught, rapidly killed with ethylacetate and brought to the laboratory for identification.

Birds were surveyed at two spatial scales, on the fields where the other species groups were surveyed and in 12.5-ha plots surrounding these fields. 'Scheme plots' were either homogeneously covered by fields with agri-environment schemes or consisted of a mosaic of fields with and without schemes. Control plots did not contain any fields with schemes. A modified approach was used in Switzerland. Farmers participating in the Swiss Ecological Compensation Area (ECA) scheme have to assign a minimum of 7% of their land as an ECA and manage it for the benefit of biodiversity. As virtually all farmers participate and average farm and field size are small in Switzerland, no 12.5-ha plots could be selected that did not contain ECA habitats. Thus, birds were surveyed at the field scale and at the 1-ha plot scale. Birds were surveyed on four occasions during the breeding season and territories were subsequently mapped following Bibby *et al.* (1992). During the breeding season most European farmland birds are territorial and territory sizes are in the range of 0.2–4.5 ha (Parish & Coulson 1998; Poulsen *et al.* 1998; Bowman 2003). Territory size may decrease with increasing suitability of the habitat (Poulsen *et al.* 1998) and farmland species may demonstrate breeding site-fidelity which tends to be stronger after a successful breeding attempt in the previous year (Thompson & Hale 1989; Groen 1993). These traits suggests that the spatial scales used in this study should be sufficient to demonstrate differences in breeding densities if the examined agri-environment schemes were having ecologically relevant effects on farmland birds.

Paired scheme and control fields were sampled for any species group on the same day and by the same person. At the end of the field season, farmers were interviewed to obtain information on the use of agrochemicals at each site. A detailed sampling protocol can be obtained from the corresponding author.

Analysis

Analyses were performed on total number of species or individuals observed per field or per 12.5-ha plot. The comparison of species richness values between sites that differ considerably in abundance has been questioned on statistical grounds and rarefaction techniques have been proposed to correct for these differences (e.g. Gotelli & Colwell 2001). However, conservation measures are aimed to increase species numbers (e.g. Hyvönen & Salonen 2002), the abundance of particular species groups or individual species (e.g. Bro *et al.* 2004) or both (e.g. Peter & Walter 2001). A diversity measure based on the number of species present per number of individuals sampled does not distinguish between responses of the number of species, individuals or both. For conservation purposes, this measure is therefore less informative than the number of species or individuals per sampling unit (i.e. species density and abundance) separately. Consequently, in this paper, we examine the effects of agri-environment schemes on species density and abundance.

Species density and abundance of Red Data Book species were analysed separately. Status of endangerment of species was obtained from national (the Netherlands and the UK) or, whenever available, from regional Red Data Books (Germany, Spain and Switzerland) corresponding with the governmental level responsible for scheme implementation. The effects of agri-environment schemes on endangered species may be difficult to detect because such species may be encountered rarely. This is not so much for birds, as many species listed as endangered are still common but are declining rapidly (for example the house sparrow *Passer domesticus* in the Netherlands and the UK). Only for plants and arthropods we therefore additionally examined the response of uncommon species (species occurring on <5% of the study fields within each country). On average 32% (SE 4.4%) of all observed species were classified as 'uncommon' species.

Effects of schemes were analysed for the different countries individually because objectives and prescriptions differed between countries. Pairs with missing or incomplete observations were omitted from the analysis even when data were missing from just one of the fields. As species density or abundance data generally do not follow a normal distribution, all species data were analysed using log-linear models employing the Poisson distribution (McCullagh & Nelder 1989) followed by a likelihood ratio test (or *G*-test) that uses a chi-square distribution. The models included 'the presence of agri-environment schemes', 'area' and nested within area, 'pair', where both area and pair were considered replications. If necessary, overdispersion was accounted for by inflating the variance of the Poisson distribution with a constant factor and assessing fit of the model using test statistics that assume *F*-distributions (Payne *et al.* 2002). The significance of effects of agri-environment schemes is indicated in figures and tables. Sample sizes and test statistics are given in Table S1.

RESULTS

In Germany, Spain and Switzerland, scheme implementation significantly reduced fertilizer and pesticide applications on the examined fields (Fig. 1). However, in Spain and Switzerland all pesticide applications consisted of spraying patches of a few problem weeds rather than full-field applications, so that ecologically significant differences in pesticide use between fields with agri-environment schemes and conventionally managed fields existed only in Germany. The absence of significant differences in agrochemical use between the two field types in the Netherlands and UK is not surprising as the use of agrochemicals is not greatly restricted by the schemes.

In all countries, except for the Netherlands, species density of plants and one of the arthropod groups was significantly higher on fields with agri-environment schemes compared with control fields (Fig. 2). In both Germany and Switzerland, bee species density was significantly enhanced on fields with agri-environment schemes. The establishment of 6-m wide grass margin strips in the UK enhanced species density of grasshoppers and crickets and in Spain the species density of spiders was raised where measures to enhance steppic birds (Table 1) were implemented. All arthropod groups that had increased species density on fields with agri-

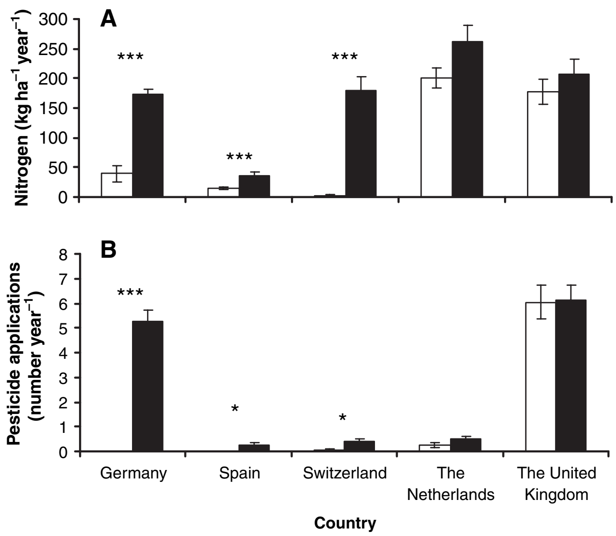

Figure 1 Use of agrochemicals on fields with agri-environment schemes (open bars) and conventionally managed fields (closed bars) in five European countries. (a) Nitrogen application. (b) Frequency of pesticide applications. Bars indicate mean values ± SE; $^{*}P < 0.05$; $^{***}P < 0.001$.

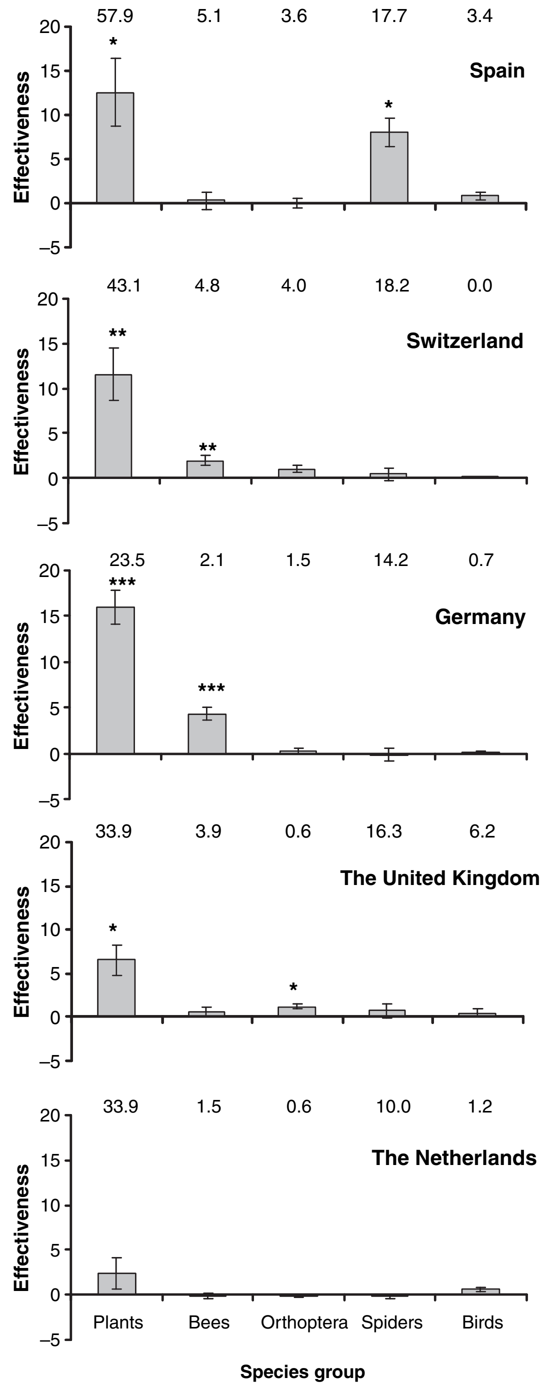

Figure 2 The effectiveness of agri-environment schemes for the conservation of five different species groups in five European countries. Effectiveness is measured as the number of species on fields with agri-environment schemes minus the number of species on paired conventionally managed fields. 'Birds' refers to territory holding birds. Bars indicate mean ± standard error of difference of mean values (SED). Numbers above bars represent mean species density on control fields. Sample sizes and test statistics are given in Table S1. $^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$.

environment schemes were also found in higher abundances (Germany, bees: $F_{1,18} = 29.6$, $P < 0.001$; Spain, spiders: $F_{1,16} = 5.42$, $P = 0.03$; Switzerland, bees: $F_{1,20} = 6.37$, $P = 0.02$; UK, Orthoptera: $F_{1,20} = 33.1$, $P < 0.001$). Species density of observed or territory-holding birds was not significantly enhanced at the field or 12.5-ha plot scale in any country ($P > 0.1$; Fig. 2; data 12.5-ha plots not shown). Species density at the field level (local diversity) was strongly related to the total number of species observed per field type within each country (regional diversity; Fig. 3). Indeed, apart from Orthoptera in Switzerland and the UK, large differences in regional diversity between scheme and control fields were only observed for species groups that differed significantly in local diversity (Table S2).

The abundance of observed birds, but not of territory-holding birds, was significantly higher at the field scale in Switzerland and Germany (Fig. 4). In Spain, birds preferentially bred on fields with agri-environment schemes, but seemed to prefer these fields for foraging to a lesser, non-significant extent (Fig. 4). In the Netherlands, more territories were observed on the 12.5-ha scheme plots consisting of a mixture of fields with postponed agricultural activities and fields with the per-clutch payment scheme (see Table 1). When comparing conventionally managed fields with fields with postponed agricultural activities only (i.e. the field scale) no significant differences were observed (Fig. 4).

The density of uncommon plant species was enhanced in Germany and Switzerland but not in Spain and the UK where positive effects on total plant species density had

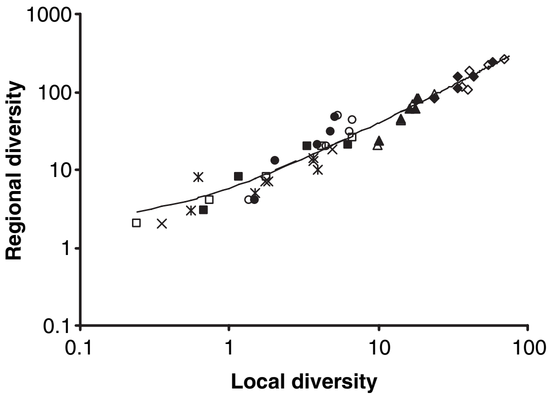

Figure 3 The relationship between local diversity (mean species density per field in each country) and regional diversity (total number of species observed on all fields). Trend line depicts result of single regression analysis on untransformed data: regional diversity = 2.03 + 3.79*local diversity, $t_{48} = 34.3$, $P < 0.001$, $R^2 = 96.4$. Bees, circles; Birds, squares; Plants, diamonds; Spiders, triangles; Orthoptera, x/asterisks. Open symbols/x, fields with agri-environment schemes. Closed symbols/asterisks, fields with conventional management. Sample sizes are given in Table S1.

been observed also (Table 2). The cover of uncommon plant species was enhanced by agri-environment schemes only in the UK. Species density of uncommon arthropods was higher on fields with agri-environment schemes in Germany and Switzerland. The abundance of these species was enhanced in Germany only. Very few of the observed plant and arthropod species were listed in Red Data Books, and only in Germany did endangered plant species occur in significantly higher numbers in fields with agri-environment schemes (Table 2). However, these comprised only 2% of the total number of plant species and their cover was not significantly enhanced (Table 2).

Endangered bird species made up a considerable proportion of the observed breeding birds in Spain, the Netherlands and the UK (83%, 44% and 28% of the observed territories respectively). However, only in Spain was the abundance of observed and territory-holding endangered birds significantly higher on fields with agri-environment schemes than on conventionally managed fields.

DISCUSSION

Effects of agri-environment schemes on biodiversity

In all countries, except for the Netherlands, some measure of biodiversity was higher on fields with agri-environment schemes compared with conventionally managed fields. Plant species density was significantly enhanced on fields with conservation management in all these countries. Increased plant species density was probably a result of reduced fertilizer and herbicide applications (Hyvönen & Salonen 2002), but may also have occurred because plant communities were buffered from the impact of agricultural activities (Moonen & Marshall 2001), such as in the UK.

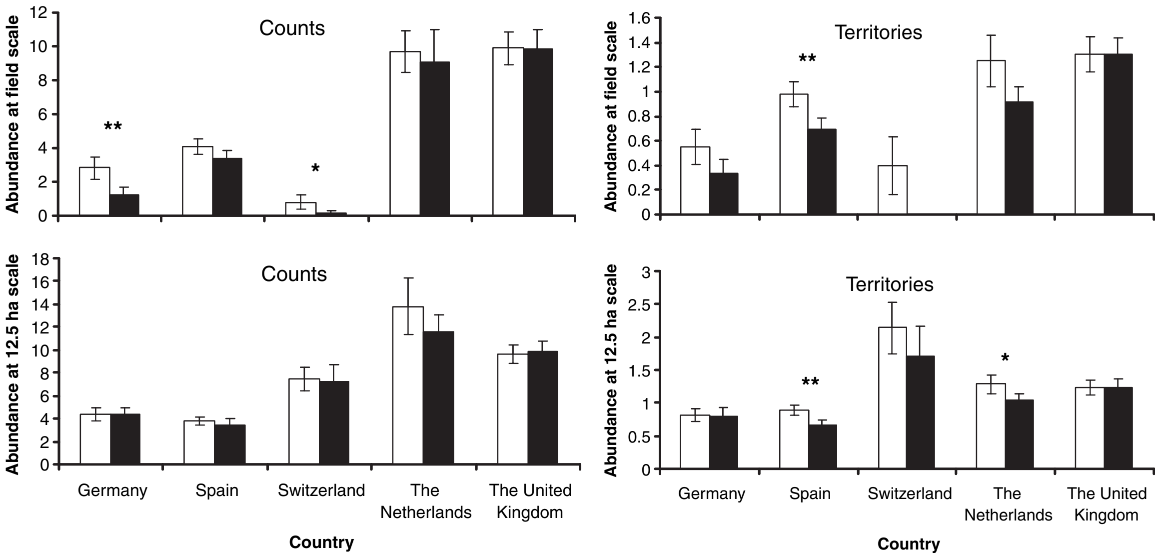

Figure 4 Mean abundance (number ha^{-1}; ± SE) of observed birds and territories at two spatial scales on agricultural land with agri-environment schemes (open bars) and paired conventionally managed land (filled bars) in five European countries. Average field size (se) in ha: Germany, 2.5 (0.49); Spain, 9.7 (0.60); Switzerland, 0.6 (0.06); the Netherlands, 1.6 (0.13); UK, 7.7 (0.84). Because of the very small size of the Swiss scheme sites, large scale plots were only 1 ha large in Switzerland. *$P < 0.05$, **$P < 0.01$.

Table 2 Mean number and abundance of uncommon species (observed on <5% of the fields within a country) and species listed in national Red Data Books on fields with agri-environment schemes (AE) and paired conventionally managed fields (control) in five European countries

	Germany		Spain		Switzerland		The Netherlands		UK	
	AE	Control	AE	Control	AE	Control	AE	Control	AE	Control
Uncommon species										
Vascular plants										
Species density	2.0*	0.8	0.9	1.5	6.4*	3.5	1.4	1.2	3.3	2.4
Cover (% total cover)	1.4	1.1	0.4	0.3	3.6	3.5	0.5	0.4	2.8*	0.9
Arthropods										
Species density	1.9*	0.7	2.2	1.5	3.1*	2.1	0	0.2	1.8	1.6
Abundance	3.2**	1.0	2.9	1.8	4.8	4.9	0	0.2	3.0	2.2
Red data book species										
Vascular plants										
Species density	0.8**	0.2	0	0	0.2	0.1	0	0	0	0
Cover (% total cover)	0.4	0.1	0	0	0.1	0.2	0	0	0	0
Arthropods										
Species density	0.2	0.1	0	0	0.9	0.7	0	0	0.3	0.1
Abundance	0.4	0.2	0	0	11.1	8.1	0	0	0.5	0.3
Birds (number ha^{-1})										
Species density (counts)	0.11	0.13	0.57	0.51	0	0	1.01	1.07	0.44	0.44
Abundance (counts)	0.30	0.36	3.61*	2.75	0	0	3.15	2.45	2.49	3.08
Species density (territories)	0.04	0.08	0.34	0.28	0	0	0.47	0.30	0.24	0.22
Abundance (territories)	0.05	0.10	0.82**	0.58	0	0	0.60	0.37	0.37	0.37

Sample sizes and test statistics are given in Table S1.
$*P < 0.05$, $**P < 0.01$.

The reduced exposure to pesticides on fields with agri-environment schemes (Fig. 1b) may have contributed to the positive response of the different arthropod groups. However, arthropod species density is often positively related to plant species density and the abundance of food resources (Siemann *et al.* 1998; Steffan-Dewenter & Tscharntke 2001), and the impact of agri-environment schemes on arthropods is probably also an indirect result of the impact of schemes on the vegetation. For example, the absence of herbicide applications on fields with schemes in Germany and the delayed earliest cut of the vegetation on fields with schemes in Switzerland probably resulted in more flowering plants, and therefore more flower-visiting insects such as bees. This is corroborated by the fact that all arthropod groups that were positively affected by schemes were found in higher abundances, suggesting a higher carrying capacity of fields with agri-environment schemes relative to conventionally managed fields. Given the significant differences in vegetation composition, and most probably vegetation structure as well, it is surprising that two of the three examined arthropod groups did not respond to scheme implementation. This may suggest that other factors, such as a small regional species pool size or landscape structure, interfered with the effects of schemes on these species groups. Interference by factors not addressed by agri-environment schemes may likewise explain the lack of positive responses of any species group to Dutch agri-environment schemes. Additionally, the general high land-use intensity in the Netherlands, even on fields with agri-environment schemes, may create environmental conditions that are only marginally suitable for farmland wildlife (Kleijn *et al.* 2001; Fig. 1).

Conservation measures may affect species composition independent of effects on species density. However, species density at the field scale was strongly positively related to the total number of species observed in a country (Fig. 3). Furthermore, positive effects of schemes on uncommon species were only observed for some of the species groups whose total species density was enhanced by schemes also. These findings suggest that species composition was not greatly altered by agri-environment schemes unless differences in species density were observed.

The absence of any differences in bird species density between fields or plots with and without agri-environment schemes is not surprising given the scale of the study. However, farmland bird abundance is known to accurately and rapidly reflect differences in resource supply (Stephens *et al.* 2003) or habitat suitability for breeding (Whittingham *et al.* 2005). In Germany and Switzerland, the observed differences between the two field types in abundance but not in territories and at the field scale but not at the 12.5-ha plot scale therefore suggests that fields with agri-environment schemes were preferentially used for foraging but that the additional resources provided by these fields may not have been sufficient to significantly increase settlement densities. In Spain, the higher densities of territories, but not of total number of observed birds, may be explained by breeding site fidelity, a trait that occurs in many farmland birds. Breeding site-fidelity often tends to be stronger after a successful breeding attempt in the previous year (Thompson & Hale 1989; Groen 1993; Arroyo *et al.* 2002). Later harvesting and the associated enhanced breeding success on fields with agri-environment schemes may thus enhance breeding densities on these fields (Arroyo *et al.* 2002). In the Netherlands, the lack of a positive response at the field level is in agreement with previous findings (Kleijn *et al.* 2001). In contrast, the modest positive effect at the 12.5-ha plot level may indicate that per-clutch-payment (Musters *et al.* 2001) is a more effective way to increase settlement densities of birds than postponing the earliest seasonal agricultural activities. The lack of any positive effect of the UK scheme on birds may be due to the fact that on average the 6-m-wide margin strips comprised *c.* 2.5% of the area of the examined fields. This quantity of newly created foraging and nesting habitat may simply not have been enough to make a difference for birds (Vickery *et al.* 2004).

Effects of agri-environment schemes on uncommon or endangered species

Uncommon species of plants and arthropods occurred in higher numbers on fields with agri-environment in Germany and Switzerland only. In these countries, part of the observed positive effect of agri-environment schemes on total species density (Fig. 2) was therefore due to species that are found only occasionally on these fields. The absence of positive effects on uncommon species in the other three countries indicates that here increases in species density were caused predominantly by common species. Positive effects of agri-environment schemes on endangered farmland species were negligible, with the exception of birds in Spain. For most species groups the low number of endangered species observed in this study prevented a reliable estimate of the effects of the agri-environmental measures. Of 14 of the 25 species groups surveyed in total in the various countries not a single individual of a Red Data Book species was observed. This confirms prior observations that contemporary farmland in north-western Europe hosts almost exclusively common wildlife species (Kleijn *et al.* 1998, 2001). Of the remaining species groups, endangered birds in Spain, the Netherlands and the UK occurred in sufficiently high numbers to reliably estimate scheme effects but only in Spain did we find positive effects of schemes. The poor effectiveness of the evaluated agri-environment schemes to promote endangered species was therefore largely because of the schemes being implemented in areas where these species simply did not occur. Additionally, where schemes were implemented in areas supporting considerable numbers of endangered species, the management restrictions were often not sufficient to enhance the population densities. This is worrisome as agri-environment schemes were introduced with the objective to counteract adverse effects of modern agriculture on the environment (Buller *et al.* 2000) and endangered species are the organisms suffering most from contemporary farming practices (Tucker & Heath 1994). From our sample, three of the schemes were targeted more or less towards rare species. The conservation of Red Data Book species is a specific objective of the Swiss scheme (Forni *et al.* 1999) whereas the schemes in Spain and the Netherlands target species groups that contain a high proportion of Red Data Book species (e.g. LNV 2003). Many of these species depend on large areas of agriculture for their persistence, and especially extensive agriculture. Therefore, protection of areas of extensive agriculture or the extensification of intensive agriculture is required rather than conservation approaches such as nature reserves that typically apply to small and/or wild areas.

Implications, limitations and perspectives

Forty-eight per cent of the examined species groups responded positively to scheme implementation, and no species group responded negatively. This is remarkably similar to the findings of Kleijn & Sutherland (2003) who found 54% of the European evaluation studies showing positive effects and 6% negative effects. As no quantifiable biodiversity objectives have been formulated for the examined schemes in any of the five countries (Table 1) the question whether the results from this study indicate that agri-environment schemes are successful in providing biodiversity benefits is open to debate. The obvious way to avoid these problems in the future and to make agri-environment schemes a more effective tool for biodiversity conservation is to formulate clear and quantifiable objectives at the start of the scheme.

This study gives an optimistic estimate of the effectiveness of agri-environment schemes. The possibility cannot be excluded that part of the positive results were due to a 'selection effect', i.e. agri-environment schemes being preferentially located on fields with high biodiversity (Kleijn & Sutherland 2003). This potential bias can only be avoided by integrating evaluation studies into agri-environment schemes at the implementation stage, so that baseline data can be collected and the preferential participation of conservation-minded farmers can be corrected for. The reluctance of most European countries to properly evaluate the ecological effects of agri-environment schemes so far suggests that this might have to be enforced, for example, by making integrated evaluation studies conditional on receiving EU co-funding for agri-environment schemes. Alternatively, countries would probably be more motivated to evaluate their agri-environment schemes if the EU would co-fund such studies rather than just provide funding for scheme implementation and not for scheme administration and evaluation (Carey 2001).

Our results indicate that the examined schemes primarily benefit common species and have limited usefulness for the conservation of endangered and often even uncommon species of farmland wildlife. Other studies have shown agri-environment schemes can be successful in promoting populations of endangered species on farmland. However, these schemes were either tailored to the needs of a single species and evaluated carefully by scientists (Evans 1997; Peach *et al.* 2001) or were located in the direct vicinity of nature reserves (Peter & Walter 2001). They represent schemes that are not very widely implemented or schemes implemented in particular, high biodiversity areas. The schemes evaluated in the current study are all widely implemented schemes and much more representative for European agri-environment schemes in general (see Table 1 in Kleijn & Sutherland 2003). This suggests that objectives of schemes may not only have to be quantified but may also have to be differentiated. Schemes aiming to increase biodiversity in general, for example with the objective to improve ecosystem processes such as pollination or pest control or to increase the value of agricultural landscapes for leisure activities, may be successful even when prescriptions are general and farming is relatively intensive. Schemes aiming to promote specific endangered species on the other hand probably need to be much more tailored to the needs of these species (Evans 1997) and need to account for environmental factors, such as dispersal barriers or groundwater level, that are outside the control of farmers but nevertheless constrain the effects of their conservation measures (Peter & Walter 2001; Kleijn & van Zuijlen 2004).

The limitation of this study approach is that for schemes that have no demonstrable positive effects it remains unclear whether this is because of the agri-environmental measures being ineffective, the implementation of measures by farmers being suboptimal, the schemes being implemented in the wrong locations or a combination of these causes. Further experimental studies are therefore required before the exact cause of the failure of agri-environment schemes to enhance biodiversity can be determined and, subsequently, scheme effectiveness can be improved. Observed positive effects, demonstrated using this study approach, suggest that schemes are implemented at a suitable location and agri-environmental measures are effective. For most European agri-environment schemes with biodiversity objectives, evidence for this is still missing (Kleijn & Sutherland 2003), and evaluation studies such as those performed here are therefore essential to determine whether the schemes are cost-effective. However, they are not enough if one wants to determine the extent to which agri-environment schemes contribute to the conservation of biodiversity at the national level. Local positive effects do not guarantee that biodiversity decline at the national or even regional level can be stopped (Berendse *et al.* 2004). This depends on effect size of schemes, e.g. whether the positive effects are strong enough to result into biodiversity increase rather than just into a reduction of biodiversity decrease, as well as upon the quantity of farmland covered by schemes (Bradbury *et al.* 2004). Additional rigorous research, and objective monitoring and evaluation is therefore required to determine whether agri-environment schemes help meet national conservation targets.

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SUPPLEMENTARY MATERIAL

The following supplementary material is available online for this article from http://www.Blackwell-Synergy.com:

Table S1 The results of statistical analyses examining the effects of agri-environment schemes on species density and abundance of various species groups. General linear models employing the Poisson distribution were used.

Table S2 The total number of species (regional diversity) observed per field type and species group in five European countries.

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