Ecological Effectiveness of Agri-Environment Schemes in Different Agricultural Landscapes in The Netherlands

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Abstract: *Agri-environment schemes are an instrument used by western European countries to counteract the negative effects of contemporary agriculture on biodiversity, but not much is known about their effectiveness. We investigated the ecological effects of Dutch agri-environment schemes aimed at promoting botanical diversity or meadow birds, and we tested whether the effectiveness of the schemes depends on landscape type or structure. In three different types of landscape, we surveyed plants, birds, bees, and hover flies on 78 paired fields that either had agri-environment schemes or were managed conventionally, and we collected data on a range of different environmental variables. Neither plant species richness nor abundance of meadow birds was higher on fields with agri-environment schemes. Landscape type had a significant effect on both species groups, but the effects of the schemes were independent of landscape type. Neither the diversity of plants nor the abundance of birds was related to any of the environmental variables. Agri-environment schemes designed to promote plant species richness or bird abundance did have positive side-effects because they enhanced the species richness of bees and hover flies, irrespective of the type of landscape. Furthermore, landscape type, groundwater level (hover flies), and area of wooded edges (bees) significantly affected both species groups. The failure of the schemes to promote the target species may be related to the high intensity of land use in The Netherlands. Simple conservation measures taken by farmers may not be sufficient to counteract the impact of factors that are often controlled at the landscape level (e.g., hydrology). Similar studies in other countries are needed to place the results of our study into a European context.*

Key Words: bees, biodiversity conservation, farmland, hover flies, meadow birds, policy evaluation, vegetation

Efectividad Ecológica de Esquemas Agri-Ambientes en Diferentes Paisajes Agrícolas en Holanda

Resumen: *Los esquemas agrícolas ambientales son un instrumento utilizado por los países de Europa occidental para contrarrestar los efectos negativos de la agricultura contemporánea sobre la biodiversidad, pero su efectividad es poco conocida. Investigamos los efectos ecológicos de esquemas agrícolas ambientales holandeses enfocados en promover la diversidad botánica o de aves de pradera, y probamos si la efectividad de los esquemas depende del tipo o estructura del paisaje. En tres tipos diferentes de paisaje muestreamos plantas, aves, abejas y moscas en 78 campos apareados que tenían esquemas agrícolas ambientales o estaban manejados convencionalmente y recolectamos datos de diferentes variables ambientales. Ni la riqueza de especies de plantas ni la abundancia de aves de pradera fueron más altas en campos con esquemas agrícolas ambientales. El tipo de paisaje tuvo un efecto significativo en ambos grupos de especies pero los efectos de los*

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esquemas fueron independientes del tipo de paisaje. Ni la diversidad de plantas ni la abundancia de aves se relacionaron con alguna de las variables ambientales. Los esquemas agrícolas ambientales diseñados para promover la riqueza de especies de plantas o la abundancia de aves tuvieron efectos secundarios positivos porque incrementaron la riqueza de abejas y moscas, independientemente del tipo de paisaje. Más aun, el tipo de paisaje, el nivel de agua subterránea (moscas) y el área de bordes leñosos (abejas) afectaron significativamente a ambos grupos de especies. La falla de los esquemas para promover a las especies focales se puede relacionar con el uso intensivo del suelo en Holanda. Las medidas de conservación simples que toman los granjeros pueden no ser suficientes para contrarrestar el impacto de factores que a menudo son controlados al nivel de paisaje (por ejemplo, la hidrología). Se requiere de estudios similares en otros países para colocar nuestros resultados en un contexto europeo.

Palabras Clave: aves de praderaabejas, conservación de biodiversidad, evaluación de políticas, moscas, tierra de cultivo, vegetación

Introduction

In densely populated Western Europe, few pristine natural areas exist. Important components of natural ecosystems, such as large grazers or predators, have been driven to extinction, and even marginally suitable land has been exploited intensively by humans. As a result, many types of seminatural ecosystems have developed that depend on human interference for their persistence but that are nevertheless characterized by high species richness. Recent developments in agriculture have led to an intensification of land use on more productive soils and abandonment of marginal land (MacDonald et al. 2000). Both trends have resulted in a dramatic decline of the biological diversity of agricultural landscapes. Consequently, many current nature conservation efforts in Western Europe aim to conserve these semi-natural ecosystems (Bignal & McCracken 1996). One approach is to create nature reserves managed according to old agricultural practices. Another approach is to convince farmers to conserve nature on their farms and to compensate them financially for any associated loss of income. The latter, called agri-environment schemes, are the focus of our paper; they have existed since 1992 and originated from European Economic Community regulation 2078/92. In a number of European countries, however, similar schemes existed well before that time. The schemes address a range of environmental problems associated with contemporary agriculture, a substantial proportion being aimed at biodiversity conservation.

Between 1992 and 2003, approximately €24 billion was spent on agri-environment schemes in the 15 countries of the European Union alone (EEA 2002). Little reliable data are available, however, on the ecological effects of agri-environment schemes. The evaluation of agri-environment schemes is hampered by the fact that most of them lack well-defined objectives (Anonymous 1998). Schemes often consist of management prescriptions that do make sense ecologically and have at times proven effective under experimental conditions (Bakker 1987; Beintema & Müskens 1987). It remains unclear, however, whether they are also effective when executed by farmers whose primary objectives are necessarily agronomic or economic rather than ecological. Furthermore, to keep the administration of the schemes manageable, in many countries a small set of conservation measures is executed in a wide variety of landscapes. Landscape structure does, however, affect the diversity of many species groups in different ways (Steffan-Dewenter et al. 2002) so that the effects of conservation measures may vary between landscapes. So far, the most convincing scientific proof of any ecological benefits of agri-environment schemes has been provided by Peach et al. (2001), who demonstrated that the Cirl Bunting (*Emberiza cirlus*) benefits from Countryside Stewardship Schemes in Devon (U.K.). This scheme was atypical in the sense that (1) it had a clearly defined target, promoting population growth of a single species, (2) it was implemented in a limited geographical area, (3) its management prescriptions were based on extensive scientific research (Evans 1997), and (4) its implementation was monitored intensively by scientists. In contrast, Kleijn et al. (2001) evaluated the effectiveness of more typical widespread schemes in The Netherlands and found that the target organisms did not benefit from the conservation measures. However, this study was carried out in a variety of landscapes scattered throughout The Netherlands, and the variability in environmental conditions may provide an explanation for the ineffectiveness of conservation measures.

We present the results of an in-depth analysis of the data of Kleijn et al. (2001). First, we aimed to determine whether the effectiveness of Dutch agri-environment schemes is affected by landscape type. Second, we wanted to find out whether the species richness of the examined species groups is related to environmental factors or types of management other than agri-environment schemes. Third, we examined whether conservation measures aimed at one species group have positive side-effects on other species groups.

Methods

Agri-Environment Schemes in the Netherlands

Contemporary Dutch agricultural landscapes are internationally important for meadow birds in general and the wader species Black-tailed Godwit (*Limosa limosa*) and Oystercatcher (*Haematopus ostralegus*) in particular. Approximately 50% and 30–40%, respectively, of the European populations of these species breed in The Netherlands (Hagemeijer et al. 1997). The most widespread form of agri-environment scheme in The Netherlands, the management agreement, has existed since 1981 (Beintema et al. 1997). Individual agreements differ in details, but all can be placed into one of the following two categories.

First, "meadow-bird agreements" prohibit changes in drainage, allow only patch-wise application of herbicides against a few problem species, and disallow activities on fields between 1 April and somewhere in June and July, depending on the agreement. The aims of meadow-bird agreements are not clearly formulated. The prescriptions are more or less tailored to enhance the reproductive success of birds, mainly wader species that use the fields for breeding (Beintema & Müskens 1987). On the other hand, population trends of a much wider set of meadow birds, including ducks (e.g., Garganey [*Anas querquedula*], Tufted Duck [*Aythya fuligula*]) and songbirds (e.g., Meadow Pipit [*Anthus pratensis*], Skylark [*Alauda arvensis*]), are usually monitored in areas with management agreements, suggesting that the responsible authorities are also interested in population trends of these species in relation to the agreements.

Second, "botanical agreements" prohibit changes in drainage, reseeding, or fertilizer application and allow only patch-wise application of herbicides. These prescriptions may apply to whole fields or only to field edges. The targets of these agreements are not formulated, but they are commonly considered to promote or conserve at least common hayfield species, thereby creating a more flower-rich vegetation.

Approach

The most powerful approach to examining the effectiveness of conservation measures on farmland would be to survey pairs of fields with and without measures from the moment of implementation and compare the development throughout the lifetime of the scheme. Unfortunately, these time series are unavailable. Because agricultural landscapes are heterogeneous, a comparison of fields or areas with and without management agreements at one point in time suffers from a large amount of environmental noise and is therefore imprecise. Furthermore, only farmers living in certain areas can enter into management agreements (similar to environmentally sensitive areas of the United Kingdom). These areas are selected for their relatively high levels of natural diversity and thus their high conservation potential. This makes a comparison of fields inside and fields outside these selected areas invalid because it would result in a bias toward fields with management agreements. Farmers usually do not have all their fields in these schemes, however, and some farmers do not participate at all. This allowed us to make a pair-wise comparison of fields with management agreements and conventionally managed fields that were both located in areas with high conservation potential. We furthermore reduced the random effects of environmental factors by selecting the two fields within each pair near each other (mean between field distance $\pm$ SE: 301 $\pm$ 169 m) so that they had the same soil type and groundwater level and were situated in similarly structured landscapes. The management agreements on the fields that were part of this study were on average 6.3 years old. Because agreements can be entered into for 6 years, effects that do not (yet) become apparent in this study can be considered ecologically irrelevant to management agreements as a nature conservation tool. We used 78 fields (39 pairs), which should provide sufficient replication to neutralize differences in initial conditions.

The 39 pairs were in nine different landscapes scattered throughout The Netherlands, of which three landscapes were on sandy soils (10 pairs), three on peaty soils (17 pairs), and three on clayey soils (12 pairs). These three landscape types host approximately 60% of all management agreements in The Netherlands and contrast sharply in soil type, openness, size and shape of the fields, and the local groundwater level. We chose the nine study areas at random from a list of areas that satisfied a few minimum requirements, such as the number of farms with management agreements and the distribution of fields with and without management agreements. This design allowed us to determine whether the impact of management agreements varied among landscape types.

Because of the high mobility of birds, we measured the effect of management agreements on birds at different spatial scales ranging from the selected fields (2.0 ha; SE = 0.18) to 12.5-ha plots around the fields. On average, 60% of the area of the 12.5-ha plots was covered by management agreements, whereas the entire 12.5-ha control plots consisted of conventionally managed fields.

Sampling of Species Groups

Because we were interested in the side-effects of conservation measures on nontarget species groups, we surveyed not only the target species groups of vascular plants and breeding birds but also bees and hover flies. Both species groups correlate well with total insect biodiversity in agricultural landscapes (Duelli & Obrist 1998). Furthermore, 194 of the 338 bee species occurring in The Netherlands are considered threatened (Reemer et al. 1999), so any additional positive effect on this species group would increase the conservation benefits of management agreements aimed at plants or birds significantly.

Because most plant species in Dutch agricultural fields are confined to the edge, on each field we surveyed plants in 20 2 × 10 m quadrats (presence data) parallel and adjacent to the ditch that usually bordered the fields. Quadrats were located 10 m apart and were at least 10 m clear from corners. We surveyed all quadrats once from 1 to 25 May 2000.

We surveyed breeding birds by means of territory mapping following Van Dijk (1996), a method broadly similar to the approach of the Common Bird Census in the United Kingdom. We visited each field five times: the successive visits were made from 22 March until 14 April (survey period 1), 17 to 28 April (2), 1 to 16 May (3), 16 to 31 May (4), and 5 to 20 June 2000 (5). Surveys were made between dawn and noon. The territory was designated as the field where the observation most indicative of a territory was made (e.g., nest, singing, or displaying males). For analysis we grouped the observed species into three functional groups typical of agricultural grasslands (results of the most common individual species are given by Kleijn et al. 2001): (1) waders (Lapwing [*Vanellus vanellus*], Black-tailed Godwit, Common Redshank [*Tringa totanus*], Oystercatcher, Common Snipe [*Gallinago gallinago*], Ruff [*Philomachus pugnax*], and Curlew [*Numenius arquata*]), (2) Meadow Pipit group (Meadow Pipit, Yellow Wagtail [*Motacilla flava*], and Skylark), and (3) Tufted Duck group (Tufted Duck, Common Teal [*Anas cracca*], Garganey, Northern Shoveler [*Anas clypeata*], and Gadwall [*Anas strepera*]). Other birds were either not ground-breeders (Bluethroat [*Luscinia svecica*], Sedge Warbler [*Acrocephalus schoenobaenus*]), and could therefore not be expected to show any response to changes in grassland management, or they were not exclusively bound to meadows and pastures in agricultural landscapes (e.g., Mallard [*Anas platyrhynchos*], Common Coot [*Fulicra atra*]). These species were grouped into a "non-meadow bird" group.

We surveyed bees and hover flies simultaneously with the belt method (Banaszak 1980). During a 15-minute walk at a constant pace through a 1-m-wide belt at the edge of the field, all observed individuals were caught and brought to the laboratory for identification (Barendregt 1991). Surveys were only made on sunny days, and the two fields within a pair were always surveyed on the same day and by the same person. We visited each field four times, the successive visits being made from 8 May to 2 June (survey period 1), 8 to 29 June (2), 11 July to 1 August (3), and 7 August to 11 September 2000 (4).

Survey of Environmental Conditions

To determine the exact nature of the management on the study fields, we distributed a questionnaire among participating farmers. Twenty-one farmers were not traceable, returned incomplete questionnaires, or declined to participate, leaving a total of 57 fields for which data on the management were available. The questions relevant to this study concerned (1) farm size and size of the cattle herd, (2) whether herbicides had been applied on the study field in 2000 (patchwise or broadscale), (3) the type and amount of fertilizer applied in 2000 to the field in question, and (4) the reason for (not) entering a management agreement on this particular field. We assumed that the year 2000 was representative for the field's recent management history. We calculated total nitrogen application per field based on the nitrogen content of the different types of manure listed by van Dijk (1999).

To determine whether biodiversity was correlated with environmental conditions besides type of management on the fields, we quantified the landscape structure in 300 × 500 m plots surrounding each field (these plots corresponded only partially with the 12.5-ha plots in which birds were surveyed). In these plots we mapped the occurrence of (1) ditches and canals, (2) areas with grassy nonagricultural vegetation (including field edges), (3) tall-grass or herbal vegetation (e.g., dominated by common reed [*Phragmites australis* (Cav.) Steudel], nettle [*Urtica dioica* L.]; nomenclature follows that of Van der Meijden 1990), (4) woody edges (field boundaries consisting of windbreaks, shrubbery, or tree lines), (5) woodlots and forests, (6) nature reserves, (7) infrastructure, and (8) buildings and farmyards. We quantified total area per category using ArcView (Environmental Systems Research Group, Redlands, California). For each field we also determined mean annual lowest groundwater level (MLGL) from various soil maps (scale 1:50,000, DLO-Staring Centrum, Wageningen, The Netherlands).

Analysis

We distinguished between management agreements aimed at birds and those aimed at plant species richness. Thus, analysis of the effects on birds was performed on a subset of field pairs from which pairs with only botanical agreements were removed. Seven fields had both meadow-bird agreements and botanical agreements; these were included in both subsets. We analyzed effects on the nontarget groups, bees and hover flies, irrespective of the type of agreement. We were particularly interested in the effects of landscape type, management agreements, and their interactions between these factors. Because the study had an unbalanced design—an unequal number of pairs nested within landscapes—we used the residual maximum-likelihood method (REML; Patterson & Thompson 1971) followed by Wald tests (GENSTAT 1993) rather than analysis of variance for all variables with error terms that were distributed normally (plants, hover flies, landscape features). In these analyses the factors of landscape and pair were considered random effects, whereas management agreement and landscape type were fixed effects. We analyzed the quantitative data of the landscape features in the same way to explore whether landscape structure varied systematically between fields with management agreements and control fields and among different landscape types.

The bird and bee data contained a high number of zero counts. These data were analyzed by means of generalized linear models (GLM) with a logistic-link function and assuming a binomial error distribution followed by a likelihood-ratio test (or *G* test). The models included the factors pair, landscape type, management agreement, and the interaction between management agreement and landscape type, where pairs were considered replications. Because only three fields with meadow-bird agreements were located in landscapes on sandy soil, analyses that tested for interactions between meadow-bird agreements and landscape type were carried out only for landscapes on clay and peat. Subsequent analyses of the effects of the main factors included landscapes on sand.

To examine which type of management or environmental factor related best with the observed species richness of the studied species groups, we performed stepwise multiple-regression analysis with forward selection of explanatory variables. Because meadow-bird agreements are not implemented in areas with low densities of meadow birds, we restricted determination of the model best describing bird diversity to field pairs that included a meadow-bird agreement and thus qualitatively good bird areas. Likewise, determination of the model best explaining plant diversity was restricted to the subset of field pairs that included a botanical agreement. We used two sets of variables. First, based on data from all study fields, we used an initial model that included all factors for which we had information and that had sufficient variation to grant an analysis (landscape, pair, management agreement, groundwater level, and the eight categories of landscape features). Second, using data from only those fields for which we had obtained information on the type of management, we used the above-mentioned initial model supplemented with the factors of nitrogen input (kg N/ha/year), farm size (ha), and cattle density (cows/ha). All proportional data were arcsin-transformed prior to analysis.

Results

Characterization of the Study Sites

We did not detect any significant differences in the landscape features surrounding fields with management agreements and fields with conventional management. There were large differences between the three types of landscape (data not shown). The total proportion of non-productive land varied between 7% in areas on peat to 24% in areas on sand. The agricultural landscapes differed mainly in amount of wooded areas ($W = 7.5$, 2 df, $p < 0.05$) and ditches and canals ($W = 26.0$, 2 df, $p < 0.001$), with landscapes on sandy soils having more wooded areas and fewer ditches and canals than the other two landscape types.

The most frequently mentioned reasons to participate in the agri-environment schemes were (1) to supplement farm income (83% of the responding participating farmers); (2) because the field was less suitable for intensive agricultural use as a result of the large distance from the farm, or of accessibility, hydrology, or shape (63%); and (3) to contribute to nature conservation (49%). The most important reasons not to participate were because (1) the management prescriptions required modifications to the farming system that were unacceptably large (50% of the responding nonparticipating farmers) and (2) the prescriptions were considered unpleasant (23%).

Fertilizer application was significantly reduced on fields with botanical agreements compared with control fields (106 and 246 kg N/ha/year respectively; t_{34}: -2.62, $p = 0.013$). A similar difference was found between fields with meadow-bird agreements and control fields (96 and 277 kg N/ha/year, respectively; t_{30}: -3.72, $p < 0.001$). Herbicide use was not different on the two field types: on about one-third of the fields farmers applied herbicides. Herbicides were usually applied once a year in a patch-wise fashion to control nettles (*Urtica dioica*), thistles (*Cirsium* spp.), and broad-leaved dock (*Rumex obtusifolius* L.).

Vascular Plants

A total of 268 mostly very common species of vascular plants was encountered on 31,000 m^2 of field edge. Only one species (*Sagina nodosa* (L.) Fenzl) found in a single quadrat was listed in the Dutch Red Data Book (Van der Meijden 1990).

Botanical agreements did not have an effect on plant species richness (Fig. 1). Their ineffectiveness was independent of landscape type (no significant interaction between management agreement and landscape type). Species richness at the field scale (400 m^2), but not at the quadrat scale (20 m^2), was significantly different between landscape types ($W = 6.8$, 1 df, $p < 0.05$). Fields on sandy soils were more species-rich than fields on clay, with intermediate values for fields on peat.

Plant species composition did not change independently of plant species richness. No differences in the abundance of the 10 most frequently encountered species were found between the two field types (Table 1). More importantly, no significant difference was found in the pooled abundance of 12 species with their optimum distribution in The Netherlands in wet hay meadows (i.e., target species; Table 1).

Linear-regression analysis did not show any significant relationship between the age of an agreement and its

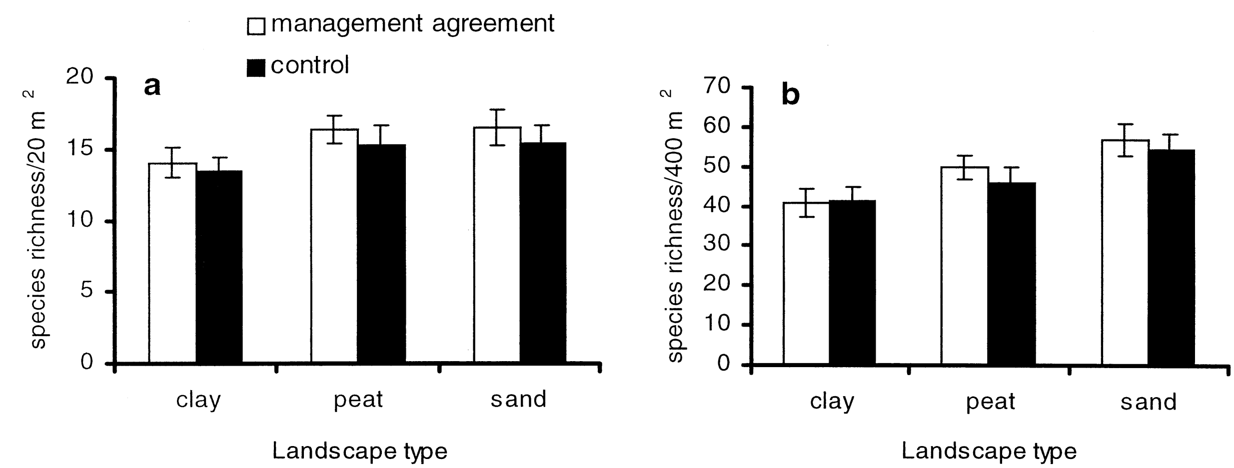

Figure 1. Effects of management agreements aimed at enhancing the botanical diversity of vascular plant species richness along edges of agricultural grasslands in different landscapes: (a) plant species richness per quadrat (±SE) and (b) plant species richness per field (sum of 20 quadrats per field) (±SE).

Table 1. Abundance (mean frequency of occurrence per field) of the 10 most frequently encountered plant species and the 12 most frequently encountered wet hay-field species in edges of fields with and without management agreements.

	Management agreements frequency		*Conventional management frequency*		
	%	*SE*	*%*	*SE*	W[a]
Common species					
Poa trivialis	93.2	2.49	94.1	2.10	0.0
Ranunculus repens	62.9	7.34	59.9	5.39	0.3
Lolium perenne	46.2	7.50	59.2	7.65	2.8
Cerastium fontanum	44.3	6.98	36.1	6.01	1.5
Agrostis stolonifera	82.2	5.59	77.0	7.65	0.1
Alopecurus geniculatus	36.5	6.30	42.0	7.51	0.6
Taraxacum officinale	48.9	5.78	55.7	6.76	0.5
Holcus lanatus	71.1	8.12	67.0	8.73	0.4
Elymus repens	56.0	7.47	50.8	7.80	0.3
Rumex acetosa	36.0	7.32	26.2	6.78	0.6
Wet hay-field species					
Ajuga reptans	13.6	7.01	14.3	7.04	
Anthoxanthum odoratum	15.9	5.10	20.3	6.70	
Cardamine pratensis	29.4	6.32	28.6	5.41	
Carex nigra	1.6	0.83	4.9	1.63	
Galium palustre	5.7	2.01	9.8	3.70	
Lotus uliginosus	7.3	3.12	6.5	3.14	
Lychnis flos-cuculi	1.8	1.17	3.9	1.86	
Lycopus europaeus	4.8	1.46	5.2	2.60	
Lysimachia nummularia	3.2	1.21	5.0	1.71	
Mentha aquatica	5.5	1.94	2.7	1.17	
Myosotis palustris	4.3	2.36	5.7	4.33	
Stellaria uliginosa	6.5	3.89	7.5	3.10	
Total wet hay-field species[b]	8.3	1.69	9.5	1.94	0.3

[a]*None of the differences was significant at* $p < 0.05$.
[b]*The wet hay-field species were not analyzed separately because most species occurred on a limited number of fields.*

effect (measured as the difference between the two fields within a pair; rc $= -0.175$, $t_{20} = -0.52$, $p = 0.61$). Multiple-regression analysis revealed that plant species richness at the quadrat level was not significantly related to any landscape or management factor (Table 2).

Birds

In general, the selected study landscapes were good meadow-bird areas. In all 20 pairs of 12.5-ha plots that contained meadow-bird agreements, exactly 500 territories of species of the three functional groups of meadow-bird species were found, resulting in an average density of 100 territories/km^2. Territories of seven threatened species were encountered: Black-tailed Godwit, Redshank, Snipe, Gargany, Black Tern (*Chlidonias niger*), Gray Partridge (*Perdix perdix*), and Sedge Warbler (Osieck & Hustings 1994).

Because no significant interactions between the effects of management agreements and landscape type were found, the effects of the two factors are presented separately. Waders were observed less frequently and had fewer territories on fields with management agreements than on conventionally managed fields (Table 3). This avoidance was not apparent at the 12.5-ha scale, but at this scale areas with management agreements did not support higher wader settlement densities than conventionally managed areas. At the 12.5-ha scale, species of the Meadow Pipit group were observed more frequently in areas with management agreements, and they tended to have more territories in these areas as well ($G = 3.63$; 1 df; $p = 0.059$). The non-meadow birds were observed in significantly higher numbers on fields with management agreements; however, this was largely due to flocks of Starlings (*Sturnus vulgaris*) that seemed to prefer to forage on these fields. Because Starlings do not breed in

Table 2. Best models explaining the richness of species groups on the study fields based on variables of landscape features surrounding the field, groundwater level, and management on the field.*

Variable	*Type relationship*	F	p
No. of plant species ($n = 44$)			
—			
No. of hover-fly species ($n = 74$)			
landscape		5.35	<0.001
management agreement	+	7.40	<0.001
mean annual lowest groundwater level	−	7.17	0.009
No. of bee species ($n = 74$)			
landscape		5.27	<0.001
management agreement	+	15.60	<0.001
area woody edges	+	4.54	0.037
		G	*p*
No. of territories, waders ($n = 40$)			
landscape		50.11	<0.001
management agreement	−	7.98	<0.01
No. of territories, Meadow Pipit group ($n = 40$)			
landscape		16.97	<0.01
No. of territories, Tufted Duck group ($n = 40$)			
landscape		16.29	<0.05
No. of territories, non-meadow birds ($n = 40$)			
landscape		23.57	<0.001
area woody edges	+	13.98	<0.001
mean annual lowest groundwater level	−	6.40	<0.05
area infrastructure	+	4.05	<0.05

**Stepwise multiple regression analysis with forward selection of variables that contributed significantly to the explanatory power of the model was used. Models for plant species richness were constructed only for field pairs with botanical agreements and models for bird species richness only for field pairs with meadow-bird agreements.*

grasslands, densities of nonmeadow bird territories were similar in the two field types (Table 3). No significant trend between the age of meadow-bird agreements and their effectiveness in increasing wader densities was found (Fig. 2).

Landscape type had a significant impact on the local bird community. Agricultural landscapes on sand were not particularly good for meadow-bird conservation (Fig. 3). However, analyses that excluded landscapes on sand showed that areas on clay also supported significantly higher densities of all four species groups than areas on peat (territories waders: $G = 314.7$, 1 df, $p < 0.001$; Meadow Pipit group: $G = 40.1$, 1 df, $p < 0.001$; Tufted Duck group: $G = 102.9$, 1 df, $p < 0.001$; non-meadow birds: $G = 204.7$, 1 df, $p < 0.001$).

Multiple-regression analysis did not reveal additional environmental or management factors that improved the fit of the models explaining territory numbers of the three meadow-bird groups (Table 2). Non-meadow birds, however, were observed in higher densities in areas with more woody edges and infrastructure (dikes, roads, railroads) and in lower densities in areas with lower mean annual lowest levels of ground water (i.e., drier areas).

Insects

We observed 70 of the approximately 300 species of hover flies that occur in The Netherlands. Only four of the observed species are considered rare (*Cheilosia rotundiventris*, *Platycheirus europaeus*, *P. immarginatus*, *P. occultus*). We found only 22 of the 338 bee species that occur in The Netherlands, three of which are considered rare (*Bombus campestris*, *B. jonellus*, *Panurgus banksianus*).

No significant interaction between the effects of management agreements and landscape type on the species richness of either hover flies or bees were observed. Although conservation measures designed to support either birds or plants did not have any positive effect on the target species groups, they did have a significant positive effect on the diversity of bees and the diversity and abundance of hover flies (Fig. 4a-c; hover-fly species richness: $G = 14.76$, 1 df, $p < 0.001$; bee species richness: $G = 19.20$, 1 df, $p < 0.001$; hover fly abundance: $G = 13.00$, 1 df, $p < 0.001$). The abundance of bees was higher on fields with conservation management than on control fields in landscapes on clay and peat but not in landscapes on sand (Fig. 4d; interaction: $G = 19.85$, 1 df, $p < 0.001$). Landscapes on sandy soils supported a more species-rich bee fauna than landscapes on peat and clay (effect of landscape type on number of bee species: $G = 38.97$, 2 df, $p < 0.001$; on bee abundance: $G = 107.79$, 2 df, $p < 0.001$), but the differences for hover flies were not significant.

There was a significant positive relationship between plant species richness in the field edge and the number of species of both hover flies and bees (number of hover-fly species: 5.92 + 0.388*number of plant species, $t_{72} = 3.12$, $p = 0.003$; number of bee species: −0.088 + 0.113*number of plant species, $t_{72} = 2.52$, $p = 0.012$). Furthermore, there was a significant relationship between the abundance of the four most frequently encountered food plants and the number of hover-fly species (Fig. 5).

Multiple-regression analysis demonstrated that, besides landscape and management agreement, the hydrology of the landscape explained an additional part of the variation in the species richness of the hover flies (Table 2). The moister parts of the landscape supported higher numbers of hover-fly species. Besides being related to landscape and management agreement, bee diversity was significantly and positively related to the area of woody edges (edges with windbreaks, shrubbery, tree lines).

Table 3. Effects of management agreements on the mean number of observed meadow birds and territories (±SE) of four functional groups.[a]

	No. observations				*No. territories*			
	ma[b]	*SE*	*cm*[b]	*SE*	*ma*	*SE*	*cm*	*SE*
Field (2.0 ha), $n = 23$								
waders	7.0***	1.54	12.0	1.89	1.3*	0.32	2.1	0.36
Meadow Pipit group	1.9	0.45	1.3	0.35	0.5	0.12	0.4	0.12
Tufted Duck group	0.3	0.12	0.4	0.12	0.2	0.11	0.3	0.09
non-meadow birds	12.6***	3.96	9.2	1.67	1.0	0.21	1.2	0.37
12.5-ha plot, $n = 20$								
waders	69.1	10.7	68.2	14.3	9.2	1.62	9.7	1.38
Meadow Pipit group	9.9**	2.67	7.7	1.72	2.6	0.67	1.8	0.45
Tufted Duck group	5.6	1.67	5.9	2.22	0.8	0.25	1.0	0.34
non-meadow birds	79.3	16.4	53.15	7.0	5.1	0.96	4.6	0.78

[a]*Probability values:* *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$.
[b]*Abbreviations: ma, fields with management agreements; cm, conventionally managed fields.*

Discussion

Vascular Plants

Management agreements did not effectively protect the diversity of vascular plants in the edges of meadows and pastures in our study areas. The lack of effect was independent of the type of landscape, and even landscape itself had a marginal effect on plant species richness (Fig. 1b). Most species encountered belong to a small set of common species that can be found in a wide range of habitats (see also Kleijn et al. 1998). These are probably the only species able to survive the local management (drainage, fertilization, herbicide use, stocking rates, harvesting methods) which results in the disappearance of historical differences in vegetation composition between areas caused by regional factors (soil type, hydrology, relief). Experimental studies show that under current conditions in Dutch agricultural landscapes it is difficult to enhance the species richness of grasslands by means of extensification. Berendse et al. (1992) found no increase in species numbers 16 years after the cessation of fertilizer inputs under an annual regime of cutting and removing vegetation. On the other hand, other experimental studies suggest that shifts in species composition can be achieved well within 6 years. Melman (1991) found an increase in wet-meadow species, such as *Lychnis flos-cuculi* L., *Anthoxanthum odoratum* L., and *Carex disticha* Hudson, on edges of agricultural grasslands after the cessation of fertilizer applications. Such shifts were absent in the present study. Furthermore, the lack of any significant relationship between age and the effectiveness of the schemes suggests that even if the contract period of these schemes extended beyond 6 years, chances of success would be slim.

The relatively high levels of fertilizer input on fields with botanical agreements were mainly the result of the inclusion of fields with agreements on edges only. Although the edges themselves were not fertilized directly, it is unclear to what extent nutrients applied to the center are transported laterally in the groundwater and thus indirectly reach the edge zone. The general lack of response of the vegetation to conservation measures may be caused by a lack of seed sources (Bakker & Berendse 1999) or by the poor dispersal of propagules in the modern agricultural landscape. For example, high stocking rates may

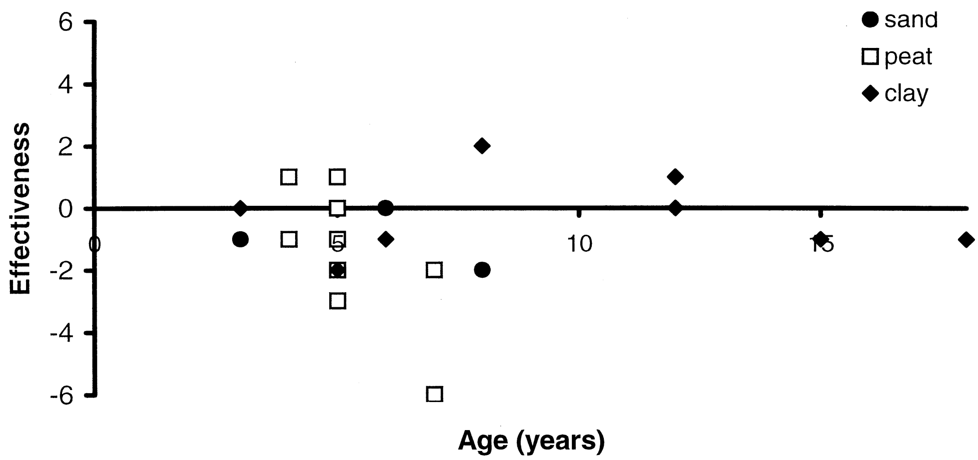

Figure 2. Relationship between age and effectiveness of management agreements aimed at supporting meadow birds. Effectiveness was calculated as the number of wader territories on the field with a management agreement minus the number of wader territories on the paired control field.

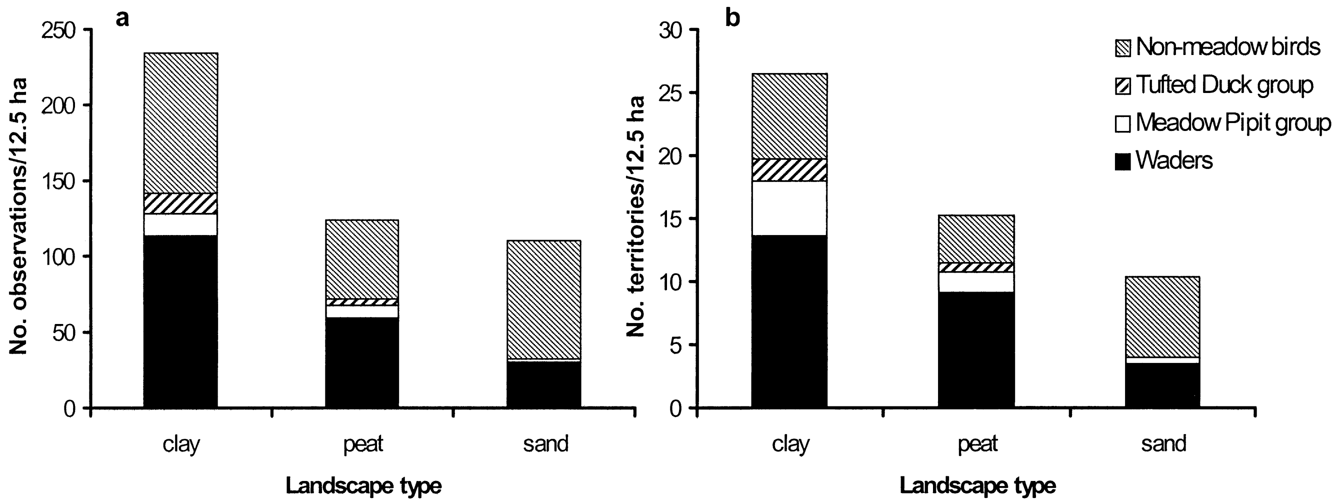

Figure 3. Mean number of (a) observations and (b) density of four functional groups of breeding birds in different agricultural landscapes (landscapes on clay, n = *10; peat,* n = *24; sand,* n = *6).*

result in the removal of seeds before they have a chance to ripen and disperse. Similarly, the shift from hay making to silaging results in an earlier mowing of the vegetation, well before seeds are ripe. Thus, even if conditions in nature-friendly managed agricultural grasslands are suitable for the establishment of new and desired species, these species may not be able to reach the site.

Birds

Meadow-bird agreements did not raise the local abundance of any of the meadow-bird groups. At the field scale, the waders, the group with the highest conservation status, avoided fields with meadow-bird agreements, although the hatching and survival rates of chicks are

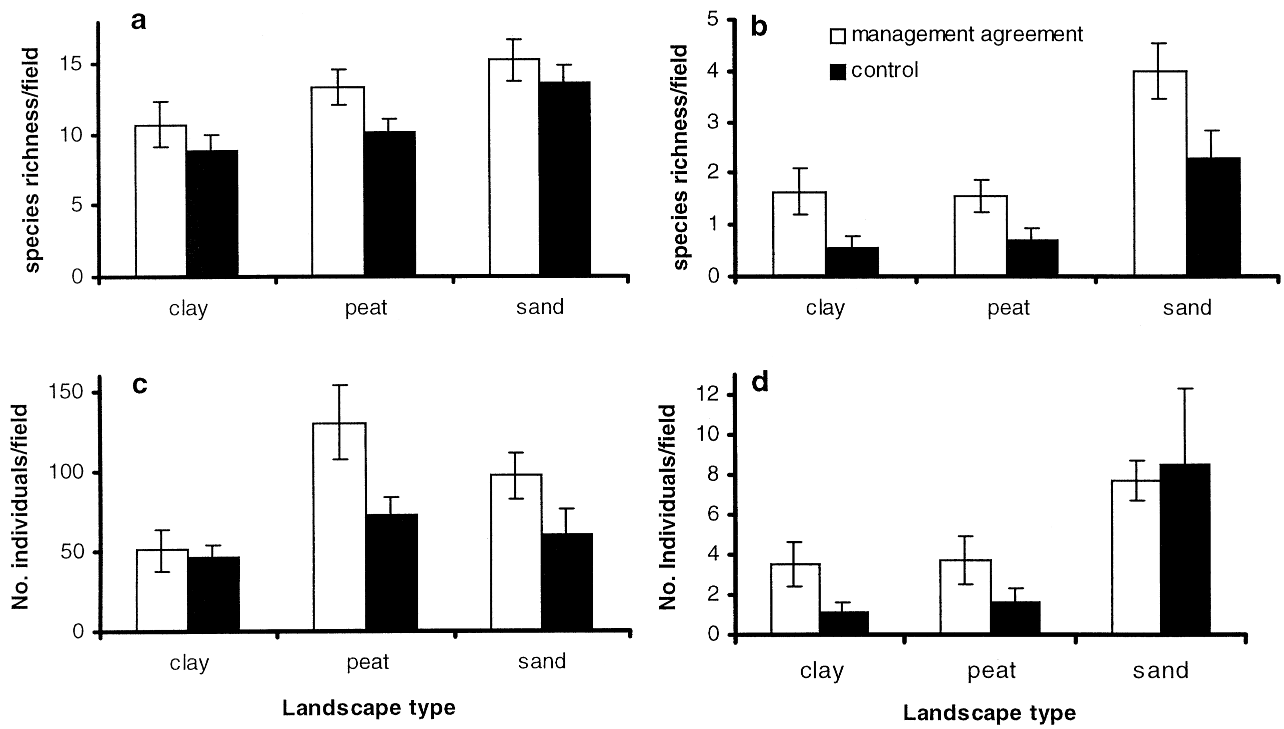

Figure 4. Effects of management agreements aimed at enhancing meadow birds and vegetation diversity on the species richness and abundance of hover flies and bees in different agricultural landscapes: (a) species richness of hover flies, (b) species richness of bees, (c) abundance of hover flies, and (d) abundance of bees (mean ± SE) (fields on clay, n = *11; peat,* n = *16; sand,* n = *10).*

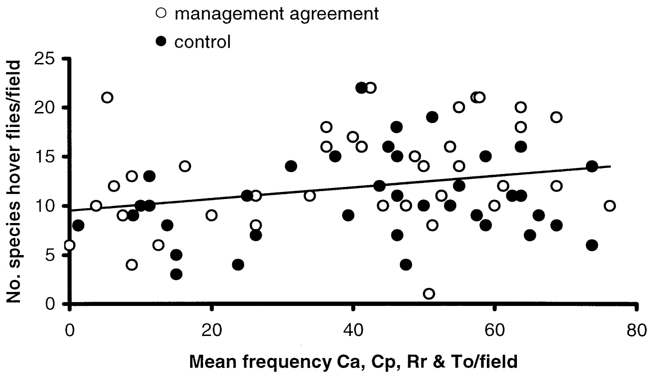

*Figure 5. Relationship between the abundance of four common plant species and the species richness of hover flies (no. hover-fly species = 9.51 + 0.0588*mean frequency;* $t_{72} = 2.23$, $p = 0.029$*) (Ca,* Cirsium arvense*; Cp, Cardamine pratensis; Rr,* Ranunculus repens*; To,* Taraxacum officinale*). Mean frequency is 100 when all four species are present in each of the 20 quadrats per field.*

enhanced on such fields (Beintema & Müskens 1987). On the 12.5-ha scale, no negative effects of management agreements were apparent. Still, neither the waders nor the Tufted Duck group was observed in higher densities in areas with management agreements, and there was only weak evidence that the Meadow Pipit group preferred these areas. Furthermore, Fig. 2 suggests that the conservation measures will not even raise wader densities in the study areas when they are being implemented well over the contract period of 6 years. The current study was carried out in a single year, and a more powerful approach of analyzing time series might have revealed effects. However, the year 2000 was a typical year for meadow birds (Teunissen et al. 2002). Furthermore, using a similar approach, Kleijn and van Zuijlen (2004) studied the effects of meadow-bird agreements on the number of meadow-bird territories in one of the study areas used in this study between 1989 and 1995. They found no differences in population trends between fields with and without agreements, which corroborates our findings.

Waders may avoid fields with management agreements because these fields receive less fertilizer and thus support lower densities of earthworms (Standen 1984), the major food items of adult waders. Waders might use the abundance of prey as an important cue in selecting their nest site, which in Dutch agricultural grasslands would result in a preference for a conventionally managed field over one with conservation management. This would result in an "ecological trap" (Kokko & Sutherland 2001). In areas with a mosaic of fields with conventional and nature-friendly management, the cues birds use to select their nesting habitat (possibly food) are no longer linked to the areas where the reproductive output is highest (areas with delayed mowing). Additionally, if the reproductive output is indeed higher on fields with management agreements, our results suggest that nest-site tenacity may not be as strong as previously assumed. Groen (1993) found that approximately 60% of the Black-tailed Godwit pairs that raised chicks successfully bred the following year within 50 m of the original nest site. A considerably lower proportion of unsuccessful pairs stayed in the same territory. In theory this would lead to higher settlement densities on fields where the reproductive success is high, such as fields with management agreements, if only because pairs tend to move away from sites with poor reproductive success, such as conventionally managed fields. However, Groen's study (1993) was carried out in an area with very high densities of birds, where most territories were occupied. Birds might not have been able to move their territory had they wanted to, unlike the birds in the landscapes we examined. It is clear that information on key ecological processes is currently missing, and the lack of a sound scientific basis might explain the poor success of the Dutch schemes.

One may argue that, due to the high mobility of birds, it is not possible to accurately allocate a territory to an individual field or 12.5-ha plot. However, Blomqvist and Johansson (1995) showed that, upon hatching, Lapwing families that bred on grasslands move on average a mere 65 ± 13 m. Furthermore, we found significant negative and positive effects on the settlement densities of birds, suggesting that the amount of replication was sufficient to compensate for noise caused by allocation errors. The aim of the measures of meadow-bird agreements is to increase the reproductive success of waders, thus enhancing population growth. We did not determine reproductive success because other researchers have demonstrated the beneficial effects of these schemes on clutch and chick survival. Nevertheless, even the increased reproductive rates of pairs on fields with agri-environment schemes seem insufficient to maintain the population size of the Black-tailed Godwitt (Schekkerman & Müskens 2000). Furthermore, there is need for concern about the net effects of these schemes because the population of the Black-tailed Godwit in The Netherlands is still declining strongly (from 85,000 in 1990 to 47,000 pairs in 2000; W. Teunissen, unpublished results), despite the establishment of these schemes in significant parts of the most favorable meadow-bird areas in The Netherlands. It is difficult to relate our results to the regional population dynamics of meadow birds. Nevertheless, our results reveal an important weakness of management agreements aimed at promoting waders: waders do not recognize these fields as superior nesting sites, and some species even tend to avoid them.

Insects

Both botanical and meadow-bird agreements had a positive side effect on nontarget species groups of hover flies and bees. The positive effect of management agreements on hover flies was mainly the result of the prolonged availability of food (nectar) on fields with a postponed mowing date (Kleijn et al. 2001). The positive relationship between hover fly diversity and the abundance of four major food plants also indicates the importance of food supply to this species group for regulating species numbers. We could not explain the positive response of bee diversity to management agreements. Our results are corroborated by the work of Kruess and Tscharntke (2002), who found that agri-environment schemes that simply reduced grazing intensity on grasslands did not result in significant changes in vegetation composition but did result in a significant increase in the species richness of a range of insect groups.

At the landscape level, the diversity of hover flies was positively related to groundwater level and bee diversity to the amount of woody edges. Both factors might be relevant to the reproductive stage of the species groups. The larvae of many common species of hover fly live in water or rotting organic material (Sommaggio 1999), both resources being more available in moist parts of the landscape than in dry parts. Many species of bees use holes in trees and shrubs to nest in (Westrich 1989). Therefore, it seems that insects can be relatively easily and rapidly promoted by supplying nest sites and food resources.

General Conclusions

Although the diversity of all investigated species groups varied significantly between landscape types, the variation in landscapes or landscape features did not explain the (in)effectiveness of Dutch agri-environment schemes. The high intensity of farming in The Netherlands may provide one explanation of why management agreements do not have the expected positive effects on target species in any of the landscapes. Over the last few decades, Dutch farmers have been encouraged to specialize and maximize agricultural output. Increased intensification has resulted in a significant modification of environmental conditions at the landscape scale. It is therefore questionable whether simple measures applied at the field scale are sufficient to conserve or restore high levels of biodiversity. Furthermore, the measures aimed at both meadow birds and vascular plants address only one of many factors that influence population growth (nutrient supply for plants and chick survival for birds). Other factors may have an overriding influence, neutralizing the effects of the agri-environmental measures.

In The Netherlands, for instance, extremely high groundwater levels and regional water seepage patterns were important factors explaining the high species richness of grasslands and the abundance of meadow birds in the past (Beintema et al. 1997). In most Dutch agricultural landscapes, groundwater levels are now kept artificially low to facilitate early access to the fields by farmers in spring, which causes the topsoil of fields to dry out more rapidly. Dried-out soils support lower densities of the soil-dwelling food items of adult waders and make them harder to reach as a result of the increased penetration resistance of the soil (Schekkerman 1997). Thus, later mowing and higher chick survival may not result in higher settlement densities of meadow birds in the long run because birds perceive the entire landscape as poor-quality nesting habitat. Generalizing this line of thought, agri-environmental measures may only be effective in areas where a minimum set of ecological prerequisites are in place. Because some important factors, such as groundwater level, are out of the control of individual farmers, schemes can best be implemented in a targeted fashion in areas where adaptations can be made at the appropriate scale: at the landscape level. Conservation measures that require little adaptations to the farming system and that are applied throughout the countryside ("broad and shallow schemes;" Curry 2002) are not likely to yield positive effects in intensively used agricultural landscapes.

It is clear that the scientific basis for the Dutch management agreements is largely missing. For example, we do not know what environmental or management factor acts as the main bottleneck controlling the population dynamics of meadow birds. Furthermore, we cannot predict the behavioral response of birds to changes in agricultural management. We need to improve our understanding of these processes before we can expect to improve the effectiveness of management agreements. The intensity of Dutch agriculture is comparable to that in certain areas in Belgium, France, and the United Kingdom (Donald et al. 2001), which is not to say that our results can be directly extrapolated to these areas. Similar problems are likely to occur in landscapes with high land-use intensity elsewhere, however, and a scientifically sound ecological evaluation of the most common agri-environment schemes is the only way to ensure that money is well spent on such schemes.

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