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# Effects of an agri-environment scheme on farmland biodiversity in Ireland

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**Abstract**

Agri-environment measures cover at least 20% of the EU's farmland, a proportion rising to approximately 30% in Ireland. A study, investigating effects on biodiversity of Ireland's Rural Environment Protection Scheme (REPS) is described. Field margin flora and Carabidae (ground beetle) fauna were surveyed on 60 paired agreement and non-agreement farms. Greater variation was observed amongst surveyed non-agreement farms: the most species-rich and species-poor farms were all non-agreement. On surveyed grassland farms, average plant species richness was significantly higher on non-agreement than on agreement farms. Otherwise, few differences between average species richness and abundance on agreement and non-agreement farms were revealed. In ordination analysis of the flora and carabidae data factors largely independent of recent management, such as hedge age and gappiness, were most important in explaining observed variation. The study concluded that the scheme has not significantly benefited the groups surveyed, and suggests that the generic measures in such horizontal schemes may be better suited to addressing landscape-level issues such as water pollution, with biodiversity objectives for high nature value areas being more effectively achieved by targeted zonal schemes. Baseline data and long-term monitoring of measurable objectives are essential for effective evaluation, both to better tailor these innovative schemes to their aims, and to clearly demonstrate their benefits.


*Keywords:* Agri-environment schemes; Monitoring and evaluation; Biodiversity assessment; Field margin; Rural Environment Protection Scheme (REPS); Carabidae

## 1. Introduction

The 2003 Mid-Term Review (MTR) of the Common Agricultural Policy (CAP) places more emphasis on rural development, the so-called Second Pillar of the CAP, an important element of which is the suite of agri-environment schemes in place across the EU (Knickel, 2002; European Commission, 2003). In 1998 one farmer in seven had an agri-environment management contract and more than 20% of EU farmland was covered by agri-environment measures (European Environment Agency, 2002). CAP Mid-Term Review

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proposals recommended an increase in the aid intensity for agri-environment programmes from 50% to 60% in the better-off regions, and from 75% to 85% in Objective 1 areas (Knickel, 2002). Furthermore, agri-environment schemes are being developed and implemented in the new EU member states, where it is hoped that they will play an important role in the mitigation of specific environmental challenges. Despite the substantial and increasing expenditure on agri-environment schemes, monitoring and evaluation have been insufficient and published data on the environmental—particularly the biodiversity—effects of agri-environment programmes are scarce (Petersen, 1998; Kleijn et al., 2001; Kleijn and Sutherland, 2003; Primdahl et al., 2003). In a comprehensive review of published studies testing the effectiveness of agri-environment schemes in protecting or enhancing biodiversity, Kleijn and Sutherland (2003) found only 62 evaluation studies originating from just five EU countries (UK, Netherlands, Germany, Ireland and Portugal) and Switzerland.

In the Republic of Ireland, agri-environment scheme coverage is higher than the EU average: in 1999, membership peaked at approximately 37% (45,553 farms, covering an area of 1,575,000 ha), although it has since fallen to about 30% (36,000 farms) (Rath, 2002). However, this is set to rise if the official minimum target of 60,000 farmers (48% of the total) in the Rural Environment Protection Scheme (REPS) by 2006 is achieved (Dept of Agriculture, Food and Rural Development, 2000).

The objective of this study was to evaluate the effects of the REPS on farmland biodiversity. The aim was to examine the impact of measures that apply to the majority of agreement farms, whether or not they are located in an area of particularly high nature value. The sampling methodology related biodiversity impacts directly to particular measures, unlike many other evaluations (Petersen and Bennett, 2001). For a discussion of recommendations for changes to the scheme that arise, see Feehan et al. (2002).

### 1.1. Biodiversity monitoring of agri-environment schemes

The wide variety of agri-environment schemes, although a good example of subsidiarity in action, mean that evaluation is a difficult task (Primdahl et al., 2003). Currently there is a small, but increasing, number of peer reviewed papers examining the impact of agri-environment schemes on biodiversity. They include the study on a Dutch agri-environment scheme by Kleijn et al. (2001), the proceedings of a British Grassland Society conference on the monitoring of grasslands in Environmentally Sensitive Areas (ESAs) in the UK (Sheldrick, 1997), a multi-disciplinary evaluation of the Countryside Stewardship Scheme (CSS) in England (Carey et al., 2003) and a methodological approach distinguishing between the performance effects and outcome effects of agri-environment schemes (Primdahl et al., 2003). Kleijn and Sutherland (2003) reviewed the literature in this area and there is a substantial quantity of so-called grey literature, including a number of important reports on the results of ESA monitoring in Northern Ireland (McAdam et al., 1994; Millsopp et al., 1997).

Although the REPS has been subject to several studies examining aspects of its ecological impact (Dunford and Feehan, 2001; Flynn et al., 2001; Aughney and Gormally, 2002), there is no system of ongoing monitoring in place. Two of these studies focused on specific regions of the country: the Burren in County Clare (Dunford and Feehan, 2001) and the Annaghmore area in County Galway (Aughney and Gormally, 2002), providing valuable information on the traditional farming practices in these regions and making recommendations for how the past can be used to inform the future in applying agri-environment measures in these areas. Flynn et al. (2001) studied the effects of REPS measures on birds, concluding that bird species richness was similar on REPS and non-REPS farms.

### 1.2. The Irish Rural Environment Protection Scheme (REPS)

Following the implementation of Council Regulation (EEC) 2078/92, the Irish Rural Environment Protection Scheme (REPS) was introduced in June 1994. The REPS is a voluntary, horizontal scheme, farmers in any part of the Republic of Ireland may apply. By contrast, some other member states have adopted a zonal approach in the development of their agri-environment schemes, whereby only farmers in certain designated areas may apply. Farmers who wish to join the REPS must undertake to implement the scheme's measures on the entire holding for five years, farming it in accordance with an individual plan which is drawn up for the farm by an approved planner.

The REPS is designed to encourage environmentally friendly farming practices by providing an incentive to retain existing sustainable farming methods, and by compensating farmers for losses incurred in making ecologically beneficial changes to their farming methods.

The main emphasis of the REPS is on nutrient management and the reduction of water pollution (Department of Agriculture, Food and Rural Development, 2000). The core measures relate to nutrient management, grassland management, protection of watercourses, retention of wildlife habitats, maintenance of field boundaries, restrictions on the use of herbicides, pesticides and fertilisers near hedgerows, lakes and streams, protection of archaeological features and visual appearance of the farm. (Further information about the REPS' objectives and measures is available in Department of Agriculture, Food and Forestry, 1994; Gillmor, 1997; Emerson and Gillmor, 1999 and Department of Agriculture, Food and Rural Development, 2000.

Of those measures applying to the majority of entrants, several are of particular relevance to the biodiversity of wild flora and fauna living on farms. Measures 3 concerns the fencing off of streamside vegetation, Measure 4 relates to retention of particular wildlife habitats, Measure 5 deals with hedgerow protection, Measure 6 concerns elimination of field margin inputs and Measure 9 includes a requirement for the widening of tillage field margins (Department of Agriculture, Food and Rural Development, 2000). It is these measures that are examined by the present study.

## 2. Methodology

Sixty farms in three counties were surveyed. Laois and Offaly are two neighbouring midlands counties which have a predominance of cattle-grazed grassland. Silage production is common, and there is a history of reclamation from bogland and subsequent soil improvement. Wexford, in the south-east, is predominantly mixed arable, with many farmers rotating between sugar beet, spring barley and winter wheat. Thirty cattle-grazed grassland farms in Counties Laois and Offaly were selected and surveyed in 1999, and 30 mixed tillage farms in County Wexford were selected and surveyed in 2000. In each farming type, 15 REPS and 15 non-REPS farms were surveyed. The REPS farms were selected randomly from shortlists of suitable farms that were compiled with the co-operation of Teagasc (the agriculture, food and development authority), who made their records available in order to facilitate the study. Selected agreement farms were paired with equal numbers of nearby non-agreement farms that were as similar as possible in every way other than membership of the scheme. In an attempt to minimise inter-farm variation, efforts were made to select clusters of a mixture of REPS and non-REPS farms.

Only farms which had been in the scheme for at least four years were examined. Although severely restrictive (less than 300 farmers entered the fifth year of the scheme in March 1999), this was necessary to ensure that any measurable effects of the scheme would have had sufficient time to manifest themselves (McAdam et al., 1994). In order to make the results as widely relevant as possible, specially designated 'high nature value' areas, such as Special Areas of Conservation (SACs) or Special Protection Areas (SPAs), were avoided.

Only certain carefully selected taxa were targeted for investigation, namely plants and carabid beetles. These taxa are potentially useful as indicator groups (Pearson, 1994; Millsopp et al., 1997); they span a wide range of guilds and trophic levels, are relatively widespread, and are responsive to alterations in management. The taxonomic stability and straightforward sampling methods for these groups are practical reasons for their selection (Pearson, 1994; Stork and Davies, 1996), and they were the two taxa focused upon by the monitoring of the ESAs in Northern Ireland (e.g. McAdam et al., 1994; Millsopp et al., 1997).

On each farm, two hedges, the field margins alongside them and one watercourse margin were surveyed. Each of these components was chosen randomly. Field and watercourse margins were surveyed using a nested quadrat system incorporating both percentage cover and species presence data. This hierarchical system of measurement increases sensitivity of the assessment to changes in the plant community (Critchley, 1997). All vascular plants present in two large 5 m × 3 m quadrats were recorded, and percentage cover data were recorded for 1 m × 3 m quadrats nested within these. The quadrats were split such that the boundary area 1.5 m out from the hedge or the watercourse was surveyed separately from the next 1.5 m band out from that, thereby enabling assessment of the impact on plant diversity of measures designed to eliminate inputs in field margins and alongside watercourses. Vegetation in the hedge was surveyed by recording all species present in the 30 m stretch. Identification followed Webb et al. (1996) and Stace (1997).

Four pitfall traps were set in each surveyed farm in early June and again in late August, giving a total of 8 traps for each farm and an overall total of 480 traps. Traps were set in the field margins, 10 m apart and 1.5 m from the hedge base. An inter-trap spacing of 10 m has been shown to optimise the number of carabid species caught from a site, while minimising damage to threatened populations (Ward et al., 2001). Identification followed Lindroth (1996) and Forsythe (1996).

Hedge height (using a clinometer), gappiness, aspect, age of watercourse fence and proximity to large non-farmland habitats such as forests, bogland or lakes were recorded from each site. Soil samples were taken from all the plant relevée sites, both from the inside 1.5 m strip and the outside 1.5 m strip, and a basic nutrient analysis was obtained. An outline of the management history and approximate input levels on the field was obtained by discussion with the farmer.

### 2.1. Data analysis

The data were examined at two scales: farm level, including farm-level averages of species richness and abundance, and regional level, for which the data from farms were pooled to allow the analysis of their collective species richness and abundance. The differences between REPS and non-REPS farms were tested using *t*-tests.

Species richness was analysed using the ordination analysis package PCOrd, which accepts presence/absence data and allows individual species and environmental variables to be singled out for individual ordination. Abundance data were analysed in the ordination package CANOCO, which accepts abundance (and percentage cover) data and allows statistical assessment of the importance of each environmental variable using Monte Carlo permutation tests. A DCA of the data was always used to ascertain that eigenvalues were sufficiently high to proceed with CCA (Ter Braak and Smilauer, 1998). Two-Way Indicator Species Analysis (TWINSPAN) was used to identify species assemblages within the data and to pinpoint indicator species for those groups.

### 2.2. Dealing with the absence of baseline data

Currently when a farm enters the REPS, no ecological baseline survey is conducted. Therefore, there is rarely information available on the species and communities present on the farm at the time of scheme entry, or on the quality of the habitats which sustained them. In the absence of ecological baseline data for REPS, non-REPS farms which were as similar as possible to selected REPS farms were surveyed. Other differences may exist between REPS and non-REPS farms which can not be factored out, but merely minimised as far as possible. This diachronic approach to biodiversity assessment remains the only option until baseline data and ongoing monitoring data are available. This method was also employed in the evaluation of the Dutch meadow birds agri-environment scheme (Kleijn et al., 2001). The present study was conducted in full awareness of the difficulties of this approach, but also with awareness of the potential of this methodology to detect outstanding impacts that the measures targeted by the assessment may be having, and point to outstanding problems in the effective implementation of those measures. It can also provide a starting point for the evaluation of the scheme and contribute to establishing a longer-term methodology for continuing that evaluation into the future, when – it is hoped – better support will be given to monitoring and evaluation.

## 3. Results

The botanical data allowed a useful comparison to be made between participant and non-participant farms. During the research a total of 160 plant species was recorded. A non-REPS grassland farm had the highest recorded number of plant species (50 species), and the most species-poor farm was a non-REPS tillage farm from which 23 plant species were recorded. Non-REPS grassland field margins were, at farm-level average, significantly more species-rich than REPS grassland margins (REPS = $12.5 \pm 3.3$, non-REPS = $14.2 \pm 3.5$, 2-sample $t$-test $p = 0.018$, alpha = 5%) (Table 1).

The results of the Carabidae survey were of considerable ecological and entomological interest, but were not as useful as the plant data in terms of REPS/non-REPS comparison. A total of 59 Carabidae species was recorded. Forty-six species were recorded on grassland, and 51 species were recorded on tillage farmland. The most species-rich and the most species-poor farms were both non-agreement farms (30 and 12 species, respectively). Species accumulation curves were calculated in order to assess the adequacy of sampling, and it was concluded from these that sampling intensity was adequate. There was no significant difference between the farm average species richness of REPS and non-REPS grassland farms, or between REPS and non-REPS tillage farms (Table 2). However, greater abundances of beetles were recorded from the wider margins in REPS tillage fields.

Relative abundance data from pitfall traps are misleading because abundance reflects activity levels rather than the actual abundance. However, abundance is of interest and importance because large numbers of predatory carabids, regardless of the species diversity of those numbers, can play a useful role in reducing crop pest numbers. In order to investigate this possibility, while taking account of the problems associated with pitfall trap abundance data, species richness was analysed separately from abundance throughout the results.

Table 1
Plant species richness in surveyed grassland and tillage field margins

| Total plant species count: 160 | No. field margins, no. farms | No. species: farm average | No. species (total)[a] |
|---|---|---|---|
| REPS grassland | 30, 15 | $12.5 \pm 3.3$ | 82 (18) |
| Non-REPS grassland | 30, 15 | $14.2 \pm 3.5$ | 78 (14) |
| REPS tillage | 28, 14 | $11.1 \pm 3$ | 81 (25) |
| Non-REPS tillage | 28, 14 | $10.8 \pm 3.8$ | 72 (16) |

[a] Number of species unique to each category is included in brackets.

Table 2
Carabidae species richness in surveyed grassland and tillage field margins

| Total Carabidae species count: 59 | No. field margins, no. farms | No. species: farm average | No. species (total)[a] |
|---|---|---|---|
| REPS grassland | 30, 15 | $19.4 \pm 6.1$ | 40 (8) |
| Non-REPS grassland | 30, 15 | $18 \pm 3.4$ | 38 (6) |
| REPS tillage | 28, 14 | $21.6 \pm 4.5$ | 44 (3) |
| Non-REPS tillage | 28, 14 | $21.4 \pm 4.1$ | 48 (7) |

[a] Number of species unique to each category is included in brackets.

### 3.1. Input reduction in field margins

Measures 4 and 6 of the REPS specify a spray limit for fertilisers, herbicides and pesticides of 1.5 m along hedgerows and streams. Only 27% of the grassland REPS farmers surveyed said that the REPS incurred a reduction in field margin inputs on their farms. On tillage farms, most REPS farmers said that they were reducing inputs in the margins and indeed P levels were significantly lower in the REPS inner field margin area (1-tailed $t$-test, $p = 0.013$).

Non-REPS grassland field margins had significantly more plant species per farm than REPS grassland margins (Fig. 1). Scaling up from farm-level to regional species richness (i.e. comparing the collective species richness for a whole category of farms, instead of comparing average farm-level species richness), the plant diversity pattern is reversed: slightly more plant species were recorded from the REPS farms (82) than from the non-REPS farms (78), with 64 species common to both groups. According to these data, the REPS does not appear to be benefiting field margin flora on surveyed farms, and in fact may be having a negative impact on field margin flora. Unnecessary fencing of field margins on REPS farms may be one reason for this (Feehan et al., 2002), whereby fences prevent the field margin from being grazed, allowing a small number of vigorous species to shade out lower-growing species. In some cases, for example to protect newly-planted quicks in the hedge, this protection from grazing is a necessary

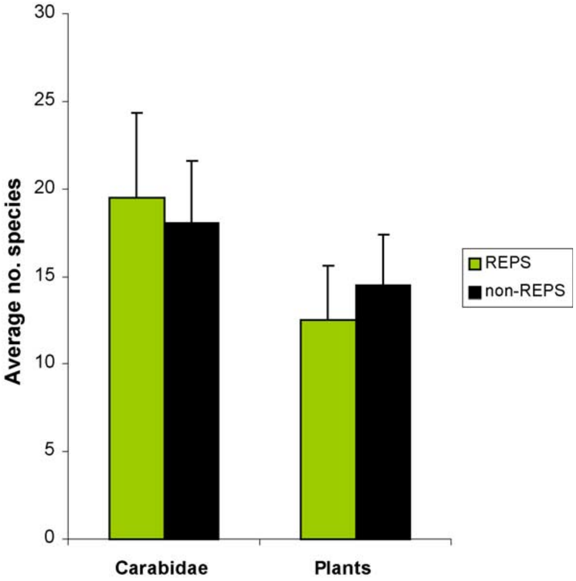


Fig. 1. Average plant and ground beetle species richness on agreement and non-agreement grassland farms ($n = 30$ farms).

step. In other cases, the fence can be set at a height and distance that allows grazing underneath without compromising stockproofing. However, longer-term monitoring would be necessary for greater understanding of the ecological effects of these management issues.

On tillage field margins, both input reduction and widening of uncultivated margins (described in the following section) have the potential to benefit flora and fauna. Plant species richness did not significantly differ between REPS ($11.1 \pm 3$) and non-REPS ($10.8 \pm 3.8$) farms (2-sample $t$-test $p = 0.66$). Scaling up, a total of 81 species were recorded in surveyed REPS tillage field margins, 72 on non-REPS margins, with 56 common to both.

Regarding beetle species richness in these field margins, averages of $19.4 \pm 6.1$ and $18 \pm 3.4$ Carabidae species were found on REPS and non-REPS grassland field margins, respectively, a difference which is not significant (2-sample $t$-test, $p = 0.44$) (Fig. 1). Overall totals of 43 carabid species from REPS grassland farms and 37 from non-REPS grassland farms were recorded, with 32 species common to both groups. The age of the hedge appeared to have an important influence on the species composition of the carabid field margin assemblage, but not on the abundance. There was a tendency for fewer species and lower abundances to be recorded alongside gappy hedges. In ordination analysis (CCA, Monte Carlo permutation test), hedge gappiness emerged as the variable explaining the most variation in the species abundance data (Fig. 2). The next most important variable was sequence, which conveyed the variation between hedges, incorporating hedge-to-hedge and farm-to-farm variation as well as the weather conditions to which earlier or later traps were inadvertently exposed.

An average of $21.6 \pm 4.5$ carabid species was recorded in the (wider) REPS tillage field margins, compared to an average of $21.4 \pm 4.1$ in the non-REPS margins, a difference which is not significant (2-sample $t$-test, $p = 0.9$). Overall, the carabid fauna on tillage farmland was found to be remarkably standard; a total of 46 carabid species were recorded from REPS margins and 49 from non-REPS margins, with 42 common to both groups.

The contrast between the margin and main field area on agreement and non-agreement farms was investigated. If the field margins on agreement farms receive lower input levels while the main body of the field receives a similar input load on both types of farm, a greater contrast between margin and field would be expected on agreement farms. This was tested by examining the contrast between plant species richness in the 1.5 m field margin area and in the 1.5–3 m area further out from the hedge. The sum of the differences was greater on the non-REPS margins, rather than on the REPS margins as had been expected. On grassland field margin species richness data, both tests yielded P-values significant at the 5% level (Table 3). The observed differences may have been due not to a reduction in margin inputs, but rather to an increase in inputs on the rest of the field. Both REPS and non-REPS field margins may have been receiving a similar (low) input load, with the input load

Table 3
Analysis of the relative differences between REPS and non-REPS grassland field margins in terms of plant species richness in the 1.5 m margin area and the 1.5–3 m area ($n = 60$ margins)

| | Sum of differences | 2-tailed $t$-test | 1-tailed $t$-test (directional) |
|---|---|---|---|
| REPS | 299 | $p = 0.0390$ | $p = 0.0195$ |
| Non-REPS | 387 | | |

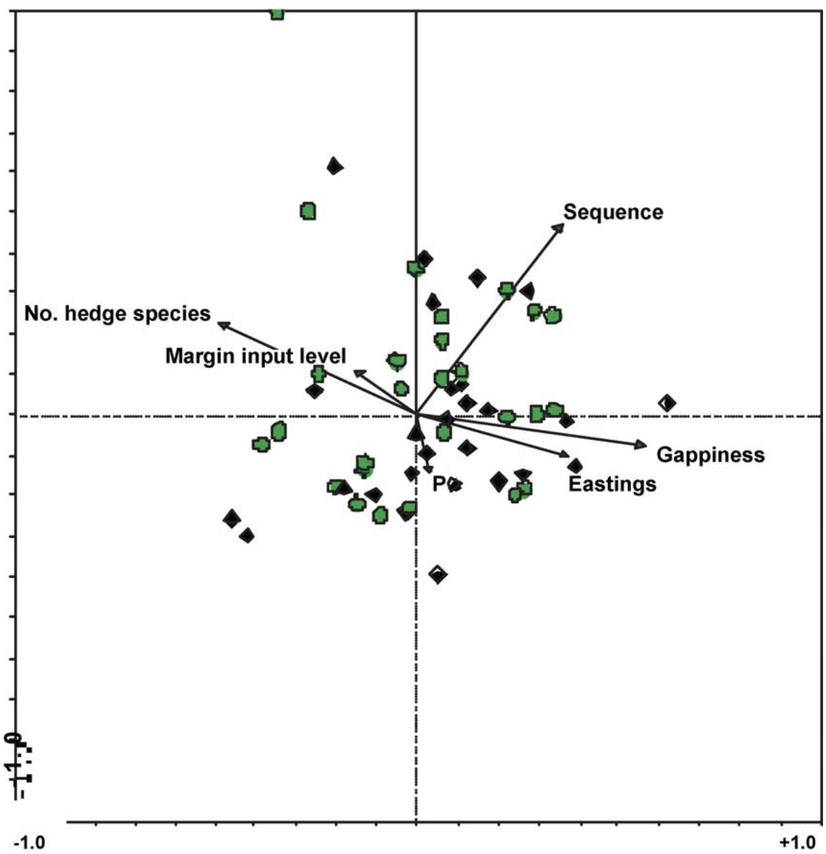


Fig. 2. Canonical correspondence analysis biplot of grassland Carabidae sites with the six environmental and management variables significant at the 5% level (Monte Carlo permutation test): hedge gappiness, pitfall trap sequence, number of species in the hedge, aspect (eastings), level of artificial inputs in the field margin and levels of P as measured in soil tests.

increasing to a greater extent on the non-REPS farms in the absence of the nutrient management restrictions that accompany the REPS. As demonstrated above, average plant species richness in surveyed grassland agreement farms was lower than on non-agreement farms, discounting the possibility that the differences between the sets of quadrats belied a greater overall species richness on REPS farms.

Overall, it was concluded that input reductions on grassland are either not being achieved in the surveyed farms, or if they are, no significant impact on floral diversity is observed. In tillage field margins, it would appear that input reductions are being achieved, and species richness in REPS margins tends to be higher than on non-REPS margins, although the difference is not significant.

### 3.2. Widening of uncultivated tillage field margins

One of the requirements of Measure 9 is that an uncultivated strip of at least 1.5 m be retained in tillage field margins. On surveyed tillage farms, REPS fields (average 181 cm) had significantly wider margins than non-REPS fields (average 145 cm) ($p < 0.001$). The surveyed REPS field margins were also significantly lower in P ($5.94 \pm 3.98$) than those of the non-REPS farms surveyed ($9.19 \pm 6.53$) (2-sample $t$-test, $p = 0.025$), whereas P-levels in the 1.5–3 m band did not differ significantly ($p = 0.25$).

These wider uncultivated field margins did not appear to significantly benefit either the plant diversity or the beetle diversity on surveyed farms (Fig. 3). This is contrary to Moonen and Marshall's (2001) finding that large boundary width contributed strongly to the diversity of hedge-bottom vegetation. However, the wider margins surveyed in that instance were several metres wide, rather than between 1.5 m and 2 m in this study. It is recommended that REPS margins be widened to at least 3 m to overcome the domination of edge effects.

In relation to the Carabidae data, it was concluded from the results that margin width showed little or no effect on carabid diversity. This is consistent with the findings of similar studies: Telfer et al. (2000) sampled the carabid fauna on arable field margins of 3 m and 6 m width, and found that margin width showed no effect on carabid assemblages. Although the wider margins did not appear to enhance beetle diversity, slightly higher abundances were recorded. This is an encouraging sign that the wider tillage margins may be providing a very real benefit by harbouring greater numbers of carabid beetles, many of which are predators of crop pests. For example, *Bembidion lampros* is an important predator of root fly eggs (Thiele, 1977). *Harpalus* and *Amara* species consume large numbers of weed seeds (Kromp, 1999), and the many polyphagous predators in the taxon contribute to the reduction of aphid populations in crops (Wratten and Powell, 1991). *Harpalus rufipes* in particular was much more abundant in the wider margins (Fig. 4(A)). It feeds on the seeds of wild plants living in the margins, and has been shown to aggregate in large numbers in suitable sites. Other species, such as the flightless, specialised snail-feeder *Cychrus caraboides*, were also more common in the wider agreement margins (Fig. 4(B)). It was found also that a significantly higher number of pygmy shrews (*Sorex minutus*) were trapped in the REPS margins (2-tailed Mann–Whitney test, $p = 0.035$). Shrews are insectivores, and their abundance reflects the availability of their invertebrate prey. Their abundance also provides an encouraging insight into the effect on a higher trophic level of widening the field margins.

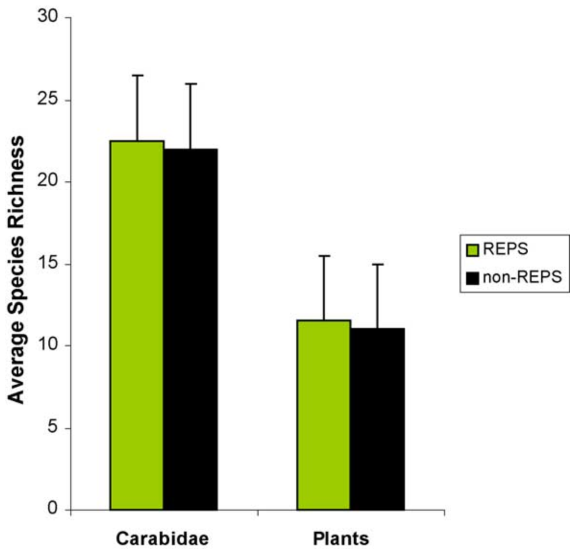


Fig. 3. Average plant and ground beetle species richness on agreement and non-agreement tillage farms ($n = 28$ farms).

### 3.3. Watercourse margin flora

REPS Measure 3 specifies that in the presence of livestock, all rivers, drains, streams and ponds must be fenced off. As well as aiming to reduce water pollution the measure aims to protect the riparian flora and allow it to develop.

However, fencing can have a negative effect on floral diversity. The mean species richness of 11 fenced REPS watercourse margins was 14.7, and that of 11 unfenced non-REPS margins was 16.1 (Fig. 5), although the difference is not significant (2-sample *t*-test, $p = 0.21$). Furthermore, it was found that the distance of the fence from the water's edge was not significantly correlated with plant margin species richness (Fig. 6). In order to factor out the influence of the REPS and to investigate the effect of fencing on the flora, fenced and unfenced watercourses within the scheme or outside the scheme were compared. In both groups, the unfenced margins had a higher number of species.

There is a clear necessity to protect large rivers and ponds from the pollution that can result from free access of stock. However, small drains that are dry for part of the year do not pose such a pollution problem. On less intensive, low-input farms, these drains and streams and their margin areas can provide a key habitat for plants and wildlife. If grazing is prevented, strong light competitors (such as woody plants and tall grasses) can grow to dominate the habitat, removing many of the low-growing plants from the habitat (Olff and Ritchie, 1998). Furthermore, the drain is prone to becoming blocked up as a result. It is

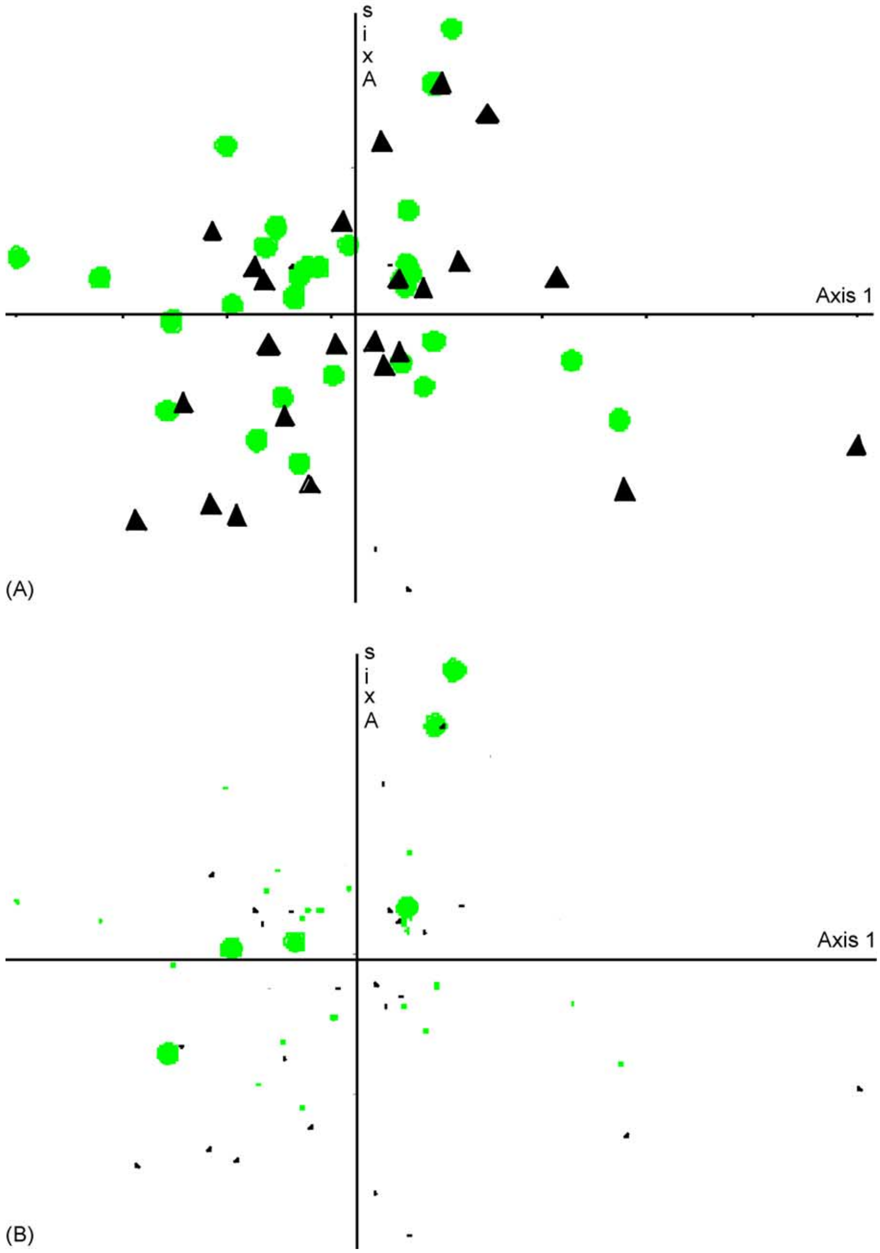


Fig. 4. (A) Scatterplot of *Harpalus rufipes*. This arable generalist was widespread on both agreement and non-agreement farms, but was recorded at higher abundances in the wider margins of agreement farms. Agreement sites are marked by circles, non-agreement sites by triangles. (B) Scatterplot of *Cychrus caraboides*, a large, flightless species that is specialised for feeding on snails. Amongst tillage farms, it was only recorded on agreement land, where it may be benefiting from the wider, less disturbed margins.

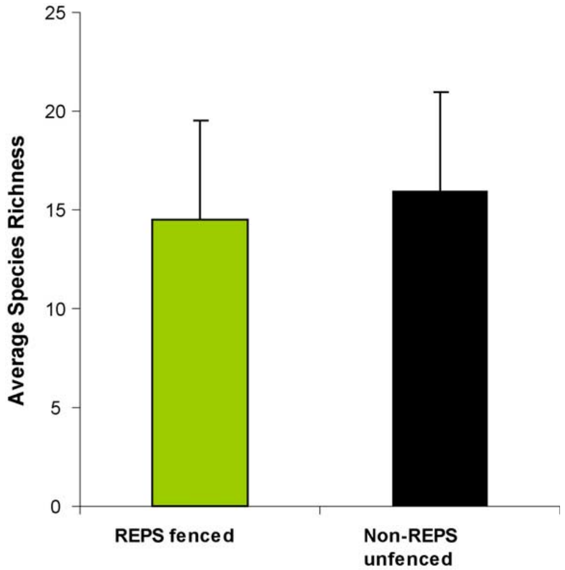


Fig. 5. The average number of plant species per farm in 11 fenced REPS watercourse margins and 11 unfenced non-REPS watercourse margins.

recommended that some of these marginal sites be left unfenced to allow these pockets of species-rich grassland and wetland vegetation to persist.

### 3.4. Hedgerow maintenance and diversity

Hedgerow species composition is a function of long-term management and species selection at the time of planting (Feehan and Keena, 2001). However, structural features of hedges such as gappiness and

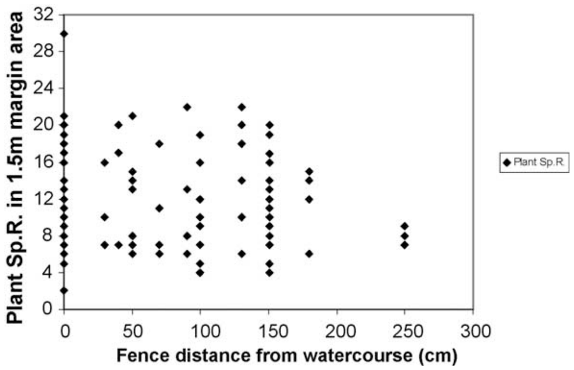


Fig. 6. The relationship between fence distance and plant species diversity in the 1.5 m margin area of watercourse margins on surveyed grassland farms ($n = 30$ farms, $R^2 = 0.0035$).

height can be influenced within a shorter time-frame. Amongst the hedges that were surveyed on grassland, it was found that fewer beetle species were recorded from alongside gappy hedges. In a tentative endorsement of the gapping-up that is being done on many REPS hedges as recommended by Measure 5 of the scheme, it appears that hedges with gaps are less effective in providing shelter and protection to beetles.

The level of hedge gappiness did not emerge as an important variable in explaining the variation observed in field margin plant species diversity in either grassland or tillage field margins. Gappiness emerged as being more important in relation to the carabid community than to the field margin flora; it was the most significant variable in explaining variation in grassland carabid abundance data (Fig. 2). Hedge age was the most important variable in explaining variation in tillage field margin plant abundance data, and it consistently emerged as an important variable in ordinations of the carabid data, particularly presence/absence. Hedge height explained a significant amount of variation observed in grassland field abundance data. The effect of the hedge's aspect appeared to be modulated by the gappiness of the hedge; aspect is less important in affecting field margin microclimate in a gappy, low hedge than in a dense, tall hedge.

## 4. Discussion

This research suggests that so far, the REPS is having little beneficial impact on the diversity of flora and surveyed fauna groups on farmland, although there were a number of encouraging signs of success relating to some individual species. Ordination analysis indicated that scheme membership explained very little of the observed variation, with other features independent of this – such as hedge age and structure – emerging as the most important variables.

The research also suggests that protection and enhancement of biodiversity requires more targeted and focused measures. Zonal schemes focusing on the preservation and enhancement of particular high nature value areas may be more effective in this regard. On the other hand, the generic measures of a horizontal-type scheme such as the REPS are well-suited to mitigating landscape-level issues such as water pollution and nutrient management.

Long-term monitoring is an essential prerequisite for a thorough understanding of ecological processes and scale effects (Hobbie et al., 2003), and also of specific ecologically-relevant policy measures such as agri-environment schemes. Britain's Countryside Survey 2000 is an excellent source of the long-term data that can underpin appropriate and achievable policy and management responses to different pressures, and the experience from CS2000 may prove helpful for the design and management of other large-scale monitoring programmes (Firbank et al., 2003). However, few countries have developed such a resource to date. Ireland is not unique in its reliance on short-term and medium-term assessments to answer questions that can only be answered by means of ongoing longer-term monitoring data. Short- to medium-term studies do not provide the continuity that is necessary to evaluate such policy measures over time, but they nonetheless afford benchmarks for future longer-term investigations.

There are good reasons why so little effective monitoring and evaluation of agri-environment schemes has been done to date: it is difficult, and it does not come cheap. However, it is worth the investment. It is recommended that it be made obligatory for each EU member state to invest a fixed proportion of its agri-environment budget on monitoring and evaluation. A small number of countries, such as the UK, have already decided to pursue this course of action. Without thorough evaluation, the future of agri-environment schemes is an uncertain one.

## 5. Conclusions

1. These results reveal few agreement/non-agreement differences between plant and Carabidae diversity on surveyed farms, although some carabid species appeared to respond well to wider field margins, and non-agreement grassland field margins had significantly more plant species per farm.
2. Ordination analysis indicated that long-term variables – such as hedge age and structure – were more important than scheme membership in explaining the observed variation.
3. Long-term monitoring is essential to improving the design of these policy measures so that they can more effectively fulfil their objectives. It is recommended that it be made obligatory for EU member states to devote a fixed proportion of the agri-environment budget to this.
4. It is suggested that protection and enhancement of biodiversity on farmland would be more effectively achieved by targeted zonal schemes than by horizontal schemes such as the REPS, whose generic measures are better suited to managing landscape-level issues.
5. Measurable, targeted objectives relating to farmland flora and fauna diversity, supported by detailed management discussion and advice, are needed.

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