Improving the Farmland Biodiversity Value of Riparian Buffer Strips: Conflicts and Compromises

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The intensity of management of lowland grassland fields in the United Kingdom, coupled with the fact that such grasslands dominate much of the lowland landscape, means that there are now few opportunities for many plants, invertebrates, birds, or mammals to survive. The Scottish Agricultural College (SAC) has investigated whether fencing off the margins of such fields next to watercourses to control diffuse pollution has any positive impacts on biodiversity, based on assessments of vegetation composition and condition and structure of assemblages of invertebrates of importance as foodstuffs to farmland birds. Fencing watercourses increased the abundance of key groups of invertebrates. However, the invertebrate species diversity was not increased unless the margins were ≥5.4 m in width. Margins established in the study area to prevent access by livestock to watercourses or to enhance biodiversity are generally ≤2.6 m wide and are therefore unlikely to provide conditions for additional invertebrate species to use. The dense, tall swards within such margins are also unlikely to provide foraging opportunities for farmland birds. Management (such as low-intensity grazing by livestock in the margins) is essential to provide the conditions required for these groups, but this could conflict with the diffuse pollution mitigation aims. A compromise is proposed whereby limited autumn/winter grazing by livestock could be used to open the vegetation structure in the margins. Grazing by livestock at that time may be acceptable since it is not occurring in the period of main diffuse pollution concern (i.e., the fecal contamination of watercourses and bathing waters in the spring and summer). It is also essential that a landscape-scale approach is taken, driven by knowledge of the full needs of the species concerned, when deciding where best to target agri-environmental actions aimed at farmland bird conservation.

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Europe's countryside and cultural landscapes have been shaped by farming over centuries (Bignal and McCracken, 1996). Farmland, including arable land and permanent grassland, is the dominant land cover in Europe, covering >45% (173 million ha) of the European Union's 27 member states (EU–27). It has been estimated that 50% of all species in Europe depend on agricultural habitats (Kristensen, 2003). However, agricultural modernization and intensification over the last 60 yr have had significant impacts on the biodiversity value of Europe's farmland. The mechanization of agriculture has facilitated the elimination of many landscape elements, such as hedgerows, drainage of wetlands, and plowing up of seminatural grasslands. Species richness and habitat diversity have also declined due to related factors such as increased pesticide and fertilizer use, simplification of crop rotations, increases in livestock grazing densities, and changes to the timing of grazing, cutting, and cropping practices (Supplemental Table S1). This development of intensively managed agricultural land has affected all agricultural sectors and has occurred across most of the lowland areas of Europe but has been especially dominant in the north and west (Henle et al., 2008).

Habitat heterogeneity is considered to be one of the most important factors (together with land use practices themselves) influencing large-scale patterns of biodiversity in agricultural landscapes (Benton et al., 2003). Many studies (e.g., Weibull et al., 2000; Schweiger et al., 2005; Hendrickx et al., 2007) have shown that increasing heterogeneity, connectivity, and area of natural and seminatural elements in an agricultural landscape tend to have a positive influence on species richness and abundance across a range of wildlife groups. There is, however, a need to ensure that these patches of seminatural habitats are not only of sufficient quality but also of sufficient size and connectivity (Whittingham, 2007). Donald and Evans (2006) suggested that restoring (or maintaining where it still exists) the agricultural landscape matrix is a necessary prerequisite to helping ensure that European agri-environmental schemes (i.e., schemes where payments are made to farmers to help them implement farming practices considered to have positive environmental benefits) fulfill their potential.

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Abbreviations: CCA, Canonical Correspondence Analysis; GLMM, Generalized Linear Mixed Model; SAC, Scottish Agricultural College; SEPA, Scottish Environment Protection Agency.

In the United Kingdom, there is increasing concern about the need to improve the overall biodiversity value of intensively managed grasslands and there is strong evidence that habitat quality for farmland birds has declined markedly throughout grassland-dominated landscapes (e.g., Robinson et al., 2001; McCracken and Tallowin, 2004). Changes in the populations of farmland birds appear to be linked to large-scale temporal changes in invertebrate numbers and seed resources (e.g., Wilson et al., 1999; Benton et al., 2002) and especially the loss of ecological heterogeneity at multiple spatial and temporal scales caused by agricultural intensification (e.g., Benton et al., 2003). The extent to which grassland management has changed in the United Kingdom over the past 60 yr is summarized by Vickery et al. (2001). The impacts on vegetation are considered in detail within that paper and within Smith (1994), whereas the impacts on invertebrates have been well documented by Curry (1994) and Morris (2000).

The biodiversity value of the wide range of river, stream, ditch, and irrigation channels occurring within farmed landscapes is closely related not only to the associated vegetation conditions at the side of these watercourses but also to farming practices and the type and condition of vegetation in the neighboring fields (e.g., Corbacho et al., 2003; Lovell and Sullivan, 2006). The ecological quality of both the watercourse and its marginal vegetation can therefore be adversely affected by farming operations either directly (through plowing) or indirectly (through runoff of livestock feces, nutrients, or pesticides applied to neighboring crops). Diffuse pollution from agricultural sources has adversely impacted the quality of water in some areas of Scotland and diffuse pollution monitoring and mitigation from rural land uses are therefore major objectives of Scotland's European Water Framework Directive strategy (SEPA, 2009a). In particular, hot spots have been identified, with many of the bathing waters along the Clyde and Solway coasts of southwest Scotland being designated as having poor water quality overall during the period 2005–2008 (SEPA, 2009a; SEPA, 2010a).

To this end, the Scottish Environment Protection Agency's (SEPA's) River Basin Management Planning program recognizes that there is a need to encourage more farmers in southwest Scotland to access diffuse pollution options through Scottish agri-environmental schemes (SEPA, 2009b). The Scottish Environmental Protection Agency plans to concentrate and target their diffuse pollution mitigation activities to a range of 14 diffuse pollution priority catchments between now and 2015 (SEPA, 2009a) and has already started detailed studies of these 14 high priority river catchments to identify pollutant sources and possible mitigation actions. These studies will form the basis of detailed plans for coordinating the work of SEPA, its partners, and other organizations in working with farmers to ensure that the appropriate diffuse pollution mitigation actions are taken (SEPA, 2010b). This catchment-targeting approach (with an additional set of diffuse pollution priority catchments being targeted between 2015 and 2021, and another set between 2021 and 2027) has the potential to focus appropriate diffuse pollution mitigation measures into each area (SEPA, 2010b), including a potentially large increase in the length of watercourses fenced off to prevent livestock access (and hence direct fecal contamination) of the water.

The establishment of buffer strips along the sides of watercourses is an accepted way of providing additional protection for the watercourse from such actions and serves to increase the diversity of the farmed landscape and provide multiple environmental benefits (e.g., Marshall et al., 2006; Lovell and Sullivan, 2006). Buffer strips also have the potential to help increase habitat heterogeneity at the farm/catchment level and thereby provide some benefit to terrestrial species of invertebrates and birds (e.g., Benton et al., 2003; Bradbury and Kirby, 2006). Field margins in intensively managed grassland support greater abundance of many invertebrates (e.g., sawfly and lepidopteran larvae, homopteran and heteropteran bugs, and predatory and phytophagous beetles) than adjacent grassland fields. These invertebrates are key dietary components for farmland birds (Cole et al., 2007; Haysom et al., 2004; Woodcock et al., 2007a; Woodcock et al., 2009). Riparian margins create continuous corridors of seminatural vegetation and thus facilitate the movement of wildlife through the landscape (Cole et al., 2008). Furthermore, as they follow the watercourse, the coordination of conservation efforts among farms is potentially easier to achieve. Riparian margins therefore have enormous potential to be multifunctional, integrating both agronomic and environmental (i.e., enhancing biodiversity and mitigating diffuse pollution) objectives on intensively managed grasslands (Muscutt et al., 1993; Cole et al., 2008).

We have been working to gain an increased understanding of the factors affecting the biodiversity value of grassland field margins, diffuse pollution buffer strips, and water margins on grassland-dominated dairy farms within the Cessnock Catchment in Ayrshire (Cole et al., 2007; 2008). This catchment has been established as a diffuse pollution priority catchment by SEPA because it represents land use patterns typical of west coast dairying and because it is impacted by diffuse pollution (McCracken, 2010). The Cessnock is a tributary to the River Irvine, which discharges at Irvine Beach, a designated bathing beach, where the condition of bathing waters has been historically poor because of the presence of agriculturally derived fecal matter in the freshwater (SEPA, 2009a). Our research has concentrated on investigating whether win-win solutions can be achieved (i.e., based on assessments of vegetation composition and condition and structure of assemblages of invertebrates of importance as foodstuffs to farmland birds, does fencing off the margins of intensively managed fields next to watercourses to control diffuse pollution have any positive impacts on biodiversity?). This manuscript provides an overview of some of the main findings from the research, indicates some of the conflicts identified, and suggests ways of managing diffuse pollution buffer strips to increase their potential to also provide wider biodiversity benefits.

Materials and Methods

Study Sites

The study focused on intensively managed grassland fields (i.e., productive ryegrass, [*Lolium perenne* L.], swards with high inputs of inorganic fertilizers encompassing livestock grazing, and/or cutting for silage) adjacent to watercourses. Study sites located on 13 grassland-dominated dairy farms in the Cessnock Catchment, Ayrshire, Scotland (55°32′38″ N, 4°21′55″ W), were studied over a 4-yr period (2006–2009). A total of 26 grassland fields adjacent to fresh watercourses were selected for study.

Within each field one to three sites (i.e., 43 sites) were established to represent a range of riparian margins that occurred within the farming landscape (Table 1). Each site was allocated to one of three categories: open sites (i.e., sites with no fence between the field and watercourse), narrow margin sites ≤2.6 m (i.e., sites with narrow fenced riparian strips, width ranging from 1.0–2.6 m, established primarily to contain livestock/mark farm boundaries), and wide margin sites ≥5.4 m (i.e., sites with wide fenced riparian strips, width ranging from 5.4–24.7 m, established with the aim of reducing diffuse pollution by preventing livestock access to the watercourse or protecting the water margin vegetation from grazing by livestock) (Fig. 1).

Figure 1 highlights that within each site, two sampling transects running parallel to the watercourse were established—one (referred to as the margin transect) adjacent to the watercourse and the other (referred to as the field transect) 4 to 6 m into the field from the margin fenceline (or in the case of open sites, 4–6 m from the margin transect). Within wide fenced margins, an additional sampling transect (referred to as the middle transect) was established at the midpoint between the fenceline and watercourse. Hence, open sites each contained two sampling transects (referred to as open margin and open field); similarly, narrow margin sites each contained two sampling transects (referred to as narrow margin and narrow field); whereas wide margin sites each contained three sampling transects (referred to as wide margin, wide middle, and wide field).

Table 1. Distribution of sampling sites across transect types and sampling year (with the number of farms at each level shown in brackets).

Year	Open sites	Narrow sites	Wide sites	Total
2006	9 (6)	6 (3)	6 (2)	21 (7)
2007	15 (10)	14 (8)	13 (6)	42 (12)
2008	14 (9)	14 (8)	13 (6)	41 (12)
2009	4 (3)	4 (2)	4 (3)	16 (4)
Total	16 (10)	16 (8)	13 (6)	43 (13)

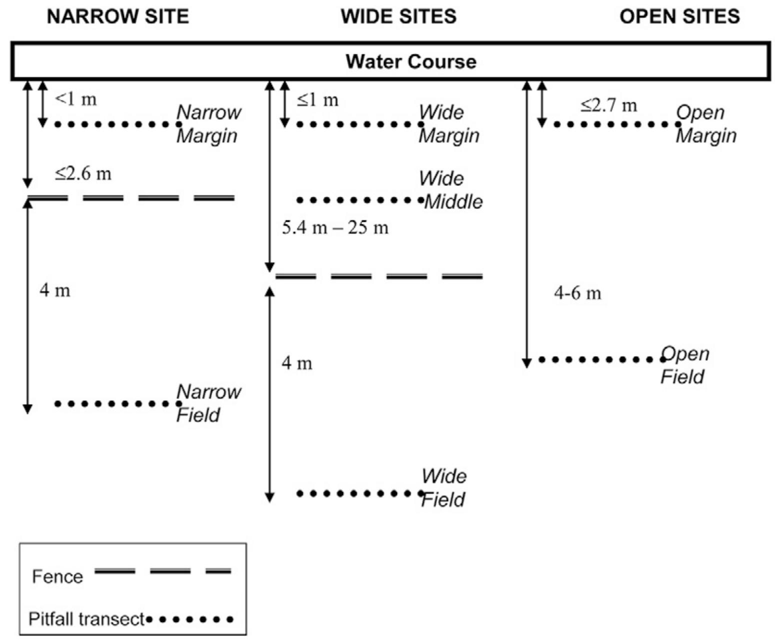

Fig. 1. Illustration of the three types of riparian zones (narrow, wide, and open) considered in the study, together with an indication of the location of the pitfall trap transects placed within each. Any one study site was characterized by only one of these three types of riparian zones.

Invertebrate Sampling

Permanent transects, each 16 m in length, were established, which formed the focus each summer of pitfall trapping to collect surface-active invertebrates together with the collection of data on other characteristics, such as vegetation assemblage structure, vegetation height, vegetation density, soil impenetrability, soil moisture, etc. (Table 2). The invertebrate data

Table 2. Environmental variables taken into consideration in the Canonical Correspondence Analyses of carabid assemblage structure indicating the *F*-value and probability value of those found to be significant via Monte Carlo Permutation tests.

Factor	*F*-value	Probability	Description
Margin width	16.0	<0.002	Margin width (m) (zero for open margin and field transects)
Vegetation height	5.87	<0.002	Mean height of vegetation throughout sampling period (cm)
Organic matter	5.39	<0.002	Soil organic matter content (%)
Wide middle	4.91	<0.002	Categorical variable: Transects in middle of wide margins
Narrow margin	4.45	<0.002	Categorical variable: Transects in narrow margins
Moisture	4.40	<0.002	Soil moisture content (%)
Wide margin	4.07	<0.002	Categorical variable: Transects in wide margins
Potassium	3.84	<0.002	Availability of potassium in the soil (mg l^{-1})
Dicotyledonous spp.	3.67	<0.002	Number of dicotyledonous plant species
Altitude	2.59	<0.002	Altitude of the pitfall trap location (m)
Impenetrability	2.52	<0.002	Soil impenetrability (lbf in^{-2})
pH	2.33	<0.002	Soil pH
Management intensity	2.13	<0.005	Management intensity of pitfall transect
Bare ground	1.76	<0.05	Percentage cover of bare ground
Distance from water	1.53	<0.1	Distance of pitfall transect from nearest water (m)
Phosphorus	1.51	<0.1	Availability of phosphorus in the soil (mg l^{-1})
Field	1.20	–	Categorical variable: All transects established in fields
Grass spp.	0.93	–	Number of grass species
Open margin			Categorical variable: Open margin transects (omitted due to collinearity)

collection concentrated on a range of taxonomic groups (such as harvestmen [Arachnida: Opiliones]; plant bugs [Hemiptera: Homoptera]; and sawfly larvae [Hymenotera: Symphyta]) known to be important as food resources for farmland birds, with ground beetles (Coleoptera: Carabidae) being identified to species level to allow detailed consideration of their assemblage structure.

At each transect surface, active invertebrates were sampled using a row of nine pitfall traps (75 mm diam. by 100 mm deep), installed at 2-m intervals to measure activity density of key invertebrate groups. Each trap contained ~50 mL of monopropylene glycol (to act as a killing agent and preservative) and was covered by a 15-mm wire mesh to reduce interference by livestock and prevent small mammals entering the trap (Cole et al., 2002). Trapping was conducted for a 4-wk period in June/July. Following collection, traps were immediately reinserted for a second 4-wk trapping period in July–August (Niemelä et al., 1990). On collection, the nine pitfall samples in each transect were pooled. As pitfall trap catches are influenced by both the density and activity of invertebrates, all analyses conducted on the counts of invertebrates are referred to as activity density (Cole et al., 2007).

Collection of Management, Soil Parameter, and Vegetation Data

Soil characteristics can directly influence not only the vegetation occurring at a location but also the types of invertebrates that can occur there, especially insects, such as ground beetles, whose larvae spend a large part of their life cycle living in soil. Hence, during pitfall installation in June, four soil cores (6 cm diam. by 10 cm deep) were taken at random from each line of pitfall traps and the soil was subjected to soil analyses used by Scottish Agricultural College's (SAC) Analytical Services Dep. to determine pH, percentage moisture content, percentage organic matter content, and phosphorus (mg L^{-1}) and potassium (mg L^{-1}) availability (SAC Analytical Services Dep., pers. comm.). Information on soil impenetrability (lbf in^{-2}) was collected using an EL29–3925 Proctor Penetrometer (ELE International, Leighton Buzzard, Bedfordshire, UK) during pitfall installation and collection. Data on the transect altitude, width of margin, and distance of transect from the water course were also collected (Cole et al., 2008).

Vegetation height was estimated using the direct measurement method (to the nearest 5 mm) with a graduated meter stick at 10 points randomly selected along each transect during pitfall installation and collection (i.e., twice during each sampling period). Mean vegetation height could therefore be calculated for each transect in each pitfall sampling period. The direct method gives consistent results and is also the preferred method of measuring vegetation heights in short swards (Stewart et al., 2001). The Robel pole visual obscurity method (Robel et al., 1970) was used to measure vegetation density. In addition, vegetation composition was determined by randomly placing four 1-m by 1-m quadrats along each transect and recording the relative abundance of plant species using the Domin scale (a standard botanical technique where a 10-point scale is used to record estimates of the percentage cover of each plant species present).

Land use and management intensity data were collected via on-site observations and annual interviews with landowners. Current and past type, and intensity of land use were derived from data collected on eight variables for each transect: sward type and age; frequency of soil disturbance, cutting, grazing, and pesticide use; and levels of inorganic and organic fertilizers applied each year. For each transect, each variable was assigned a score on a four-point scale, from 0 to 3, in ascending order of intensity. For example, for soil disturbance, a score of 0 was given for no soil disturbance in the previous 3 yr; a score of 1 indicated only harrowed once in the previous 3 yr; a score of 2 indicated only plowed once in the previous 3 yr; whereas a score of 3 indicated plowed twice or more in the previous 3 yr. For cutting, a score of 0 was given for no cutting of vegetation in the previous 3 yr; a score of 1 indicated that the vegetation had only been topped in the previous 3 yr (i.e., cutting the top of tall grass stems to prevent seed set and hence maintain consistent grass quality and growth through the grazing season); a score of 2 indicated one complete cut and removal of vegetation in the previous 3 yr; whereas a score of 3 indicated two or more complete cuts and removal of vegetation in the previous 3 yr. Once the scores were recorded for all eight variables, a composite management intensity index for each transect was calculated by summing the relevant individual variable scores. The management intensity index therefore had a potential range of between 0 and 24, with higher values indicating a greater overall intensity of agricultural management (Downie et al., 2000; Cole et al., 2002), though in practice the maximum observed on these study sites was a management intensity score of 19.

Data Analyses

Before all analyses, data were log or arcsin square root transformed to normalize where required. Generalized Linear Mixed Models (GLMMs) applying Residual Maximum Likelihood were used to analyze activity density of key invertebrates per pitfall trap (i.e., sawfly larvae, harvestmen, and homopteran bugs), soil properties (i.e., pH, moisture content, organic matter content, available phosphorus, and potassium and impenetrability), and vegetation density. The GLMMs enabled a hierarchy of random effects to be incorporated within the model and thus enable a greater strength of comparison between transects on a specific site and sampling date. With the exception of vegetation density (where vegetation height was omitted from the model), the following model was applied:

Random effects: farm + field + site + pitfall transect + annual sample

Fixed effects: year (i.e., 2006, 2007, 2008, 2009) + riparian treatment (open margin, open field, narrow margin, narrow field, wide margin, wide middle, wide field) + vegetation height.

The ecological requirements of many ground beetle species are well known and hence the structure of the ground beetle assemblage at any location cannot only provide information on the relationships between the beetle assemblage and vegetation type but also can be used to suggest the likely current and recent past vegetation structure, general environmental conditions, and management occurring at that location (Niemelä et al., 1990; Cole et al., 2002). To this end, Canonical Correspondence Analysis (CCA) (ter Braak and Šmilauer, 2002) was conducted on the ground beetle species data (combined for the two annual sampling dates) to determine the main environmental factors driving ground beetle assemblage structure. The CCA was conducted, without downweighting rare species, on the species relative abundance data rather than absolute abundance, a procedure that facilitates the standardization of pitfall sampling effort in different habitats (Cole et al., 2008).

Fourteen continuous environmental variables and four categorical variables were considered for analysis (Table 2). A fifth categorical variable (open margin) was omitted due to collinearity. To minimize problems associated with multicollinearity, a forward selection process was applied and only variables found to be statistically significant (at the 5% level) by the Monte Carlo Permutation test were included in the analysis (ter Braak and Šmilauer, 2002).

Results

General Soil and Vegetation Characteristics

Average soil potassium levels (with means varying from 69.5 ± 6.5 mg l^{-1} in wide margin treatments to 133.4 ± 7.9 mg l^{-1} in fields) and phosphorous levels (with means varying from 4.0 ± 0.5 mg l^{-1} in narrow margin treatments to 6.2 ± 0.9 mg l^{-1} in wide middle treatments) were generally moderate across all the treatments, with the observed ranges generally falling between the very low to high categories (SAC Analytical Services Department, pers. comm.). Average soil pH varied a little across the five treatments (with the field treatment mean being the lowest at 5.5 ± 0.03 and the wide margin treatment mean being the highest at 6.1 ± 0.09), whereas average soil moisture content and soil organic matter levels, and especially the observed ranges, showed more variation across the five different treatments. For example, the widest range in soil moisture content was observed in the field treatments (from 9.6–54.3%) and the lowest range in the wide middle treatments (from 10.0–41.7%), whereas the widest range of soil organic matter content was observed in the field treatments (from 4.1–45.2%) and the lowest in the open margin treatments (from 1.6–12.4%). Soils in the fields adjacent to the riparian margins and in those margins that were unfenced were more compacted, with on average a greater force being necessary to insert the penetrometer to a standard depth (means of 45.9 ± 1.6 lbf in^{-2} and 47.6 ± 2.9 lbf in^{-2} in the field and open margin treatments, respectively, compared with 27.1 ± 1.1 lbf in^{-2} in the wide margin treatments). The vegetation in riparian margins fenced off from the adjacent fields was generally taller and denser (e.g., mean height of 45.9 ± 3.2 cm and mean density of 21.9 ± 1.6 cm in wide margin treatments) than that in either the fields themselves (mean height of 11.3 ± 0.6 cm and mean density of 8.5 ± 0.3) or in those riparian margins that were unfenced (mean height of 16.9±2.0 cm and mean density of 9.0±0.8) (Supplemental Table S2). Supplemental Table S2 provides details of the means and observed ranges for all the soil and vegetation measurements across the five treatments.

Effects of Year, Treatment, and Vegetation Height on the Invertebrate and Soil Parameters

The GLMM analyses were used to investigate the relationships between these variables in more detail. Table 3 provides a summary of the influence of year, treatment, and vegetation height on the activity density of key invertebrate groups, soil parameters, and vegetation density. With the exception of soil available phosphorus, all the other variables investigated showed significant variation among sampling years ($P < 0.01$ for sawfly larvae and $P < 0.001$ for each of the other variables).

Significant influences of riparian treatment were found for the three invertebrate groups investigated ($P < 0.001$ each for harvestmen, homopteran bugs, and sawfly larvae). As can also be seen in Fig. 2, the activity density of all three invertebrate groups was greater in all the fenced margin sites (i.e., narrow margin, wide margin, wide middle) than in field sites (i.e., open field, narrow field, wide field) or open margin sites. There was no influence of margin width on harvestmen or sawfly larvae, but the activity density of homopteran bugs was greater in wide middle sites than in wide margin or narrow margin sites (indicating that the activity density of homopteran bugs was greater in the middle of the wide margins than it was close to the watercourse).

Table 3. Results of Generalized Linear Mixed Model (GLMM) showing influence of year, treatment, and vegetation height on abundance of key invertebrates, soil properties, and vegetation density. Table displays *F*-values and direction of effect when applicable for vegetation height, numerator degrees of freedom (ndf), denominator degrees of freedom (ddf), positive (+ve).

	Year ndf = 3 ddf = 163.2–181.4	Treatment ndf = 6 ddf = 67.8–89.9	Vegetation height ndf = 1 ddf = 161.9–240.7
Harvestmen	22.0***	15.77***	7.8** (+ve)
Homopteran bugs	14.9***	23.03***	2.53
Sawfly larvae	5.19**	8.6***	0.77
Soil organic content	23.4***	16.74***	0.17
Soil moisture content	62.9***	2.39*	2.22
Soil available phosphorus	0.6	0.92	2.38
Soil available potassium	4.18***	1.67	0.02
Soil impenetrability	96.1***	18.04***	1.09
Soil pH	6.68***	8.17***	0.9
Vegetation density	12.01***	26.92***	–

* $P \leq 0.05$.

** $P \leq 0.01$.

*** $P \leq 0.001$.

With the exception of soil available phosphorus and potassium, significant treatment effects were found for all the soil parameters investigated (Table 3). Soil moisture content ($P < 0.05$) was similar between the field sites and fenced margin sites, but the average and observed range of soil moisture content in the open margin sites was lower than for any other treatment. Soil organic matter content ($P < 0.001$) tended to be higher in the fields than the adjacent margins and the wide middle sites generally had higher soil organic matter content than the wide margin sites. Soil impenetrability ($P < 0.001$) was greatest in the field sites and open margin sites than within the fenced margin sites. However, vegetation density ($P < 0.001$) showed the opposite trend and was greater in the

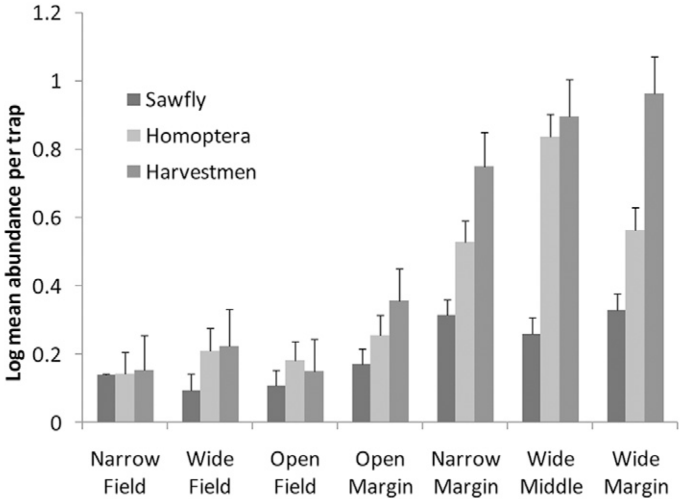

Fig. 2. Mean abundance (log) ± standard error (SE) of sawfly larvae, homopteran bugs, and harvestmen in pitfall traps across the five treatments.

fenced margins than the open margins or fields. Significant treatment effects were also found for pH ($P < 0.001$), with the margin sites (whether fenced or unfenced, with the exception of narrow margins) tending to have higher pH than adjacent fields and this difference being particularly marked between the wide margin and wide field sites.

Vegetation height was only found to significantly influence harvestmen ($P < 0.01$) with higher activity densities of harvestmen in longer grass. It is, however, important to note that the treatments that were fenced were ungrazed by livestock and as such the effects of treatment and vegetation height are confounded.

Effects on Ground Beetle Assemblage Structure

Over the 4-yr sampling period, 22,284 carabids consisting of 55 species were collected by pitfall trapping. Canonical Correspondence Analysis of the data yielded eigenvalues of 0.3430, 0.2216, 0.1451, and 0.1216, accounting for 6.3, 4.1, 2.6, and 2.2% (for axes 1–4, respectively) of the total variation in carabid assemblage structure.

The resultant ordination indicated that treatment was a major factor determining the separation of sites in the ordination space (Fig. 3). It can be seen that the assemblage structure of ground beetles in field and open margin sites were similar to each other (as these sites are clustering toward the left-hand side of the plot) and assemblage structure in the vast majority of the narrow margin sites was not markedly different from the field or open margin sites. Only in wide margin and wide middle sites does the ground beetle assemblage structure differ markedly from the other sites studied (as is indicated by the former sites lying far to the right of the plot).

Of the 18 environmental variables examined, 14 were found to have a highly significant impact on carabid assemblages (Table 3) and these variables accounted for approximately 21.6% of the observed variation in carabid assemblage structure. Vegetation height ($P < 0.002$), margin width ($P < 0.002$), soil impenetrability ($P < 0.002$), and management intensity ($P < 0.005$) were the principal factors driving the separation of the carabid assemblages and associated sites along axis 1. In agreement with findings of GLLMs, field sites were found to have the most impenetrable soil and lowest vegetation height. They were also found to be subjected to the highest intensity of management. Vegetation height, dicotyledon plant species richness ($P < 0.002$), and soil moisture content ($P <$

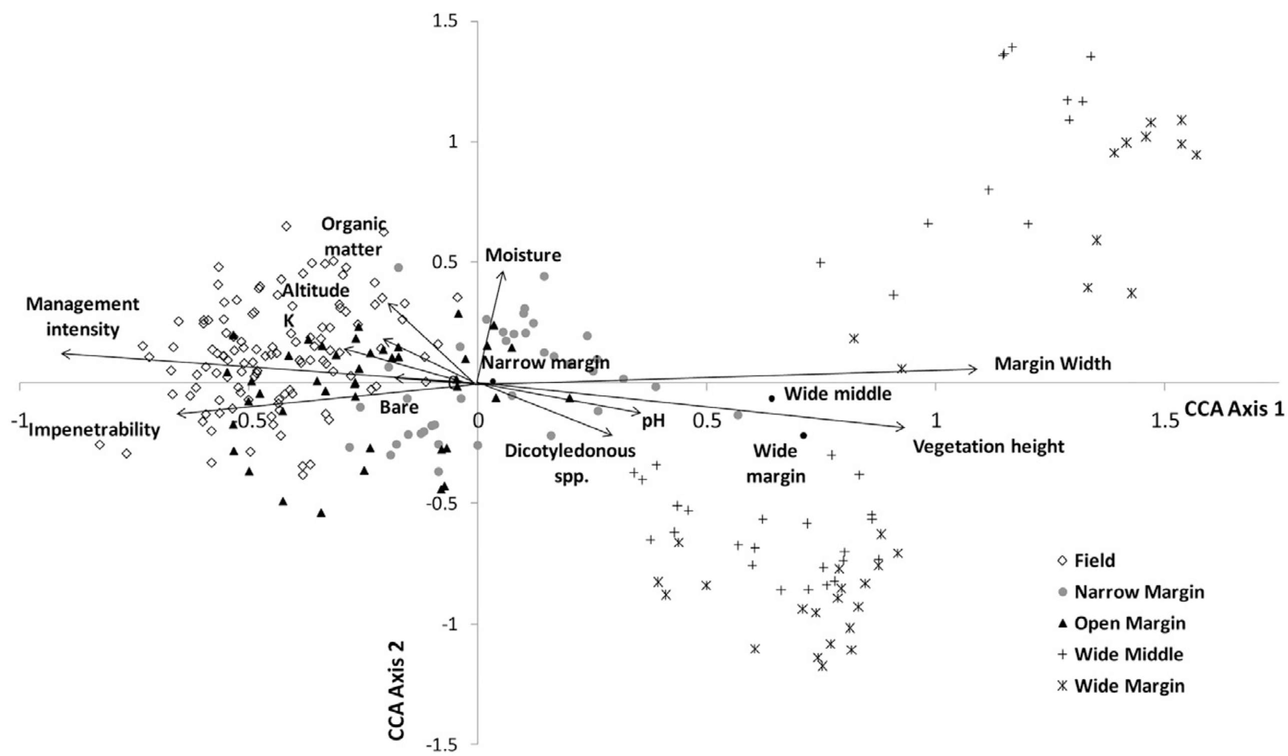

Fig. 3. Biplot derived from Canonical Correspondence Analysis of the carabid relative abundance data highlighting the different treatments, continuous environmental variables (vectors), and categorical environmental variables (dark, closed circles). Only environmental factors significant at the 5% level are included in the figure (Table 3).

0.002) appeared to have a strong influence on the separation of the sites to the right of Fig. 3, with the relatively drier wide margin and wide middle sites occurring to the bottom right of the ordination plot having shorter vegetation and more dicotyledonous plant compared with the wetter wide margin and wide middle sites occurring to the top right of the ordination.

Discussion

Some of the soil parameters assessed were found to differ between the fenced riparian margins, unfenced riparian margins, and adjacent fields. In particular, soil compaction was much less in the fenced margins and these sites also contained less soil organic matter, particularly in the fenced wide margins adjacent to the watercourse, when compared with the field sites. Soil characteristics can influence directly not only the vegetation occurring at a location but also the types of invertebrates that can occur there, especially insects, such as ground beetles, whose larvae spend a large part of their life cycle living in the soil. Hence, the observed differences in underlying soil characteristics between the fields and margins would help explain in part the observed differences in ground beetle assemblages present at the different locations. In addition, dicotyledon plant species richness was found to influence carabid assemblage structure. Hence, the greater dicotyledon plant species richness in wide margins could be an additional factor explaining why carabid assemblages in wide margins differed markedly from narrow margins, as additional carabid species were able to take advantage of the different microclimate and foraging opportunities associated with the more species-rich vegetation (Cole et al., 2008).

It is also interesting that soil organic matter content in fenced narrow margins and in the middle of wide margins was similar to the adjacent fields, possibly suggesting that fencing off field margins may not protect them from overspill or surface runoff arising from slurry spreading in adjacent fields. The much lower soil organic matter levels close to the watercourse in the unfenced margins and wide-fenced margins could potentially indicate a greater observance of the restrictions on applying slurry close to watercourses when the watercourse can be clearly seen (i.e., along unfenced margins) and greater diffuse pollution mitigation arising from fenced margins being wide (i.e., application of any slurry up to the fence in the neighboring field would mean slurry still gets into narrow margins and as far as the middle of wide margins). However, the fact that none of the margins showed any difference from the field in terms of soil-available phosphorous or potassium would suggest that nutrients are still entering the riparian margins. Although there were differences in soil pH levels among the treatments, these differences would not be expected to have any adverse impact on potassium and phosphorus (and nitrogen, though this was not assessed) availability to plants in the margins (EFMA, 2006). Southwest Scotland is a high rainfall area and this, coupled with the permanent nature of grassland in field sites, means it is not surprising that there was little difference in soil moisture content between margins and adjacent field sites.

Vegetation in fenced margins was denser and taller than in adjacent fields or unfenced margins. Ungrazed vegetation not only tends to be botanically richer in terms of the number of plant species present but also architecturally more diverse, supporting a greater array of plant structures (i.e., stems, leaves, flowers, grass tussocks, and seed heads) (Morris 2000; Woodcock et al., 2009). Indeed, Woodcock et al. (2007b) found that both vegetation assemblage structure and architectural complexity both played a key role in determining the diversity of phytophagous (such as sawfly larvae and homopteran bugs in our study) and predatory invertebrates (such as harvestmen and ground beetles in our study). Both the taxonomic and architectural complexity of the ungrazed margins are likely to have benefitted directly the three invertebrate groups investigated (i.e., harvestmen, sawfly larvae, and homopteran bugs), resulting in higher activity densities of these invertebrates in fenced margins.

While fencing off field margins increased activity density of invertebrates that would otherwise occur in very low numbers in the unfenced riparian margins of intensively managed grasslands, only when margins were ≥5.4-m wide did differences in carabid assemblage structure occur. Margins established in the study area to prevent livestock access to watercourses or that have a primary focus on biodiversity enhancement (through agri-environmental payments) are generally ≤2.63 m wide and therefore unlikely to enhance the diversity of carabid assemblages occurring at the farm and wider landscape scale. Some margins experienced periodic disturbance from irregular overflows of the watercourse, which helped open the vegetation structure and reduce competition from grasses (which otherwise benefit from nutrient loads in soil due to historic intensive management of margins before they were fenced), thus potentially making these margins more accessible to foraging birds. However, in most cases, narrow margins were situated on steeper banking than wide margins and thus did not experience flooding. Hence, when left unmanaged/undisturbed, the tall, dense, grass-dominated swards within these narrow margins are likely to provide very little in terms of increased foraging opportunities for farmland birds.

The Need for Management of the Riparian Margins

Many of the invertebrate groups that benefit from the exclusion of livestock from riparian margins (e.g., plant bugs and sawfly larvae) provide food for foliage-gleaning birds, such as the yellowhammer (*Emberiza citrinella*) (Bradbury et al., 2000). As these prey items were scarce in the adjacent grassland fields, riparian margins might be expected to have the potential to increase food resources for foliage gleaners. A greater abundance of invertebrates, however, does not necessarily mean richer foraging grounds for farmland birds, as both accessibility and detectability of prey must also be considered (Vickery et al., 2001). The tall, dense vegetation that was typical of the ungrazed margins would be expected to reduce the accessibility and detectability of invertebrate prey and impede the movement of the foraging bird (McCracken and Tallowin, 2004).

The conditions required by the plants, insects, and farmland birds currently under pressure on grassland-dominated farms are intimately linked to grazing or cutting management practices (Bignal and McCracken, 1996). Consequently, simply fencing off riparian margins can never solely redress farmland biodiversity declines. Some form of management is essential if those margins are to provide conditions required for these target groups. The best and most obvious management to use for open vegetation would be low-intensity grazing, since this produces a range of high and low vegetation (and hence opportunities for other plants and insects to colonize the margins [McCracken and Bignal, 1998; Pykälä, 2003]). This does, however, potentially conflict with the diffuse pollution mitigation aims of such margins, since these are intended to stop livestock access to (and hence fecal contamination of) watercourses with the consequent downstream impacts on bathing water quality.

It is clear that a win-win solution, in terms of achieving both diffuse pollution control and positive biodiversity benefits, cannot be achieved without management of riparian margins. So, we propose a compromise whereby limited autumn/winter grazing by livestock (preferably cattle, since they cope better with grazing longer vegetation [Dennis et al., 2008]) of wide margins (≥5.4 m to maximize biodiversity return) would help provide a more open vegetation structure in the margins the following spring/summer. This is based on the premise that grazing by cattle in autumn/winter may be an acceptable compromise since it is not conflicting with the bathing season. The concept that grazing opens vegetation is based on sound ecological principles and experience (e.g., Hayes and Holl, 2003; Pykälä, 2003; Schaich et al., 2010) but would need to be proven in the case of diffuse pollution mitigation strips (not just to show that biodiversity benefits could be achieved but also that the diffuse pollution role of the margins was not compromised).

The Wider Landscape Context

Landscape simplification is the key driver of farmland biodiversity declines (e.g., Benton et al., 2003; Hendrickx et al., 2007). To help redress such simplification, intensive grassland farmers are coming under pressure to increase the amount and type of other habitats occurring on their farms and incorporate wider environmental goals (such as for biodiversity and diffuse pollution) into their farm management practices. To limit the overall amount of farmed land that needs to be taken out of production to address these goals, there is need to ensure that multiple environmental benefits can be achieved as much as possible from the establishment of any new habitats at the farm and wider landscape levels. Riparian margins clearly have a potential role to play in increasing habitat diversity and thereby have the potential to help intensive grassland farms achieve positive diffuse pollution and biodiversity benefits while also remaining agriculturally productive.

However, the biodiversity value to be gained from any one margin, field, or habitat is also strongly influenced by its surroundings (e.g., McCracken and Bignal, 1998; Weibull et al., 2000). Hence, any assessment of the potential biodiversity impact of changes at a field scale must also take into account what changes in the type and distribution of other land covers may be necessary in the agricultural landscape. Therefore, there is a need to take into account the ecological processes and drivers influencing farmland bird utilization of grassland landscapes at a scale (such as a whole farm or suite of farms) much greater than an individual riparian margin or field. Even if riparian margins are established and managed appropriately to provide good access to abundant invertebrate food during the breeding season to a bird such as the yellowhammer, other resource requirements must also be addressed within the agricultural landscape, such as suitable nesting sites in close proximity to foraging locations (e.g., Bradbury et al. 2000) and seed-rich habitats to provide food in winter (e.g., Siriwardena et al. 2000). Such a landscape-scale approach driven by the needs of the targeted grassland bird species is essential in judging the likely impact of broad land-use changes and the choice of best locations to target agri-environmental actions aimed at grassland birds conservation.

Conclusions

To date, many agri-environmental actions in Scotland, as in many other parts of Europe, have been targeted solely at an individual environmental issue (be it diffuse pollution mitigation or biodiversity conservation) and implemented at the level of an individual field (or smaller). Many other agri-environmental actions have been targeted solely at individual farming practices (e.g., livestock grazing densities, slurry spreading) or individual components in the landscape (e.g., arable crops, grassland, woodland, hedgerows, riparian margins) considered in isolation from each other. The danger of such a restricted approach is that opportunities to achieve the maximum possible biodiversity and wider environmental benefits can be lost through the lack of coordinated planning and action. The importance of the wider landscape and the need to obtain multiple environmental benefits therefore has to be taken into account much more within the development of agri-environmental schemes (Hopkins et al., 2007). From the particular perspective of farmland birds, it will also be essential to move away from the perception that grassland birds can only be influenced by management changes directed at grassland-based habitats (Robinson et al., 2001; Perkins et al., 2000; Siriwardena et al., 2000). The importance of interactions among (and contributions arising from) other farmed and nonfarmed habitats need to be taken into account much more when seeking to enhance the value and attractiveness of intensively managed grasslands to farmland birds (McCracken and Tallowin, 2004).

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References

Benton, T.G., D.M. Bryant, L. Cole, and H.Q.P. Crick. 2002. Linking agricultural practice to insect and bird populations: A historical study over three decades. J. Appl. Ecol. 39:673–687.

Benton, T.G., J.A. Vickery, and J.D. Wilson. 2003. Farmland biodiversity: Is habitat heterogeneity the key? Trends Ecol. Evol. 18:182–188.

Bignal, E.M., and D.I. McCracken. 1996. Low-intensity farming systems in the conservation of the countryside. J. Appl. Ecol. 33:413–424.

Bradbury, R.B., and W.B. Kirby. 2006. Farmland birds and resource protection in the UK: Cross-cutting solutions for multi-functional farming? Biol. Conserv. 129:530–542. doi:10.1016/j.biocon.2005.11.020

Bradbury, R.B., A. Kyrkos, A.J. Morris, S.C. Clark, A.J. Perkins, and J.D. Wilson. 2000. Habitat associations and breeding success of yellowhammers on lowland farmland. J. Appl. Ecol. 37:789–805.

Cole, L.J., D.I. McCracken, L. Baker, and D. Parish. 2007. Grassland conservation headlands: Their impact on invertebrate assemblages in intensively managed grasslands. Agric. Ecosyst. Environ. 122:252–258. doi:10.1016/j.agee.2007.01.016

Cole, L.J., D.I. McCracken, P. Dennis, I.S. Downie, A.L. Griffin, G.N. Foster, K.J. Murphy, and T. Waterhouse. 2002. Relationships between agricultural management and ecological groups of ground beetles (Coleoptera: Carabidae) on Scottish farmland. Agric. Ecosyst. Environ. 93:323–336.

Cole, L.J., R. Morton, W. Harrison, D.I. McCracken, and D. Robertson. 2008. The influence of riparian buffer strips on Carabid beetle (Coleoptera, Carabidae) assemblage structure and diversity in intensively managed grassland fields. Biodivers. Conserv. 17:2233–2245. doi:10.1007/s10531-007-9304-1

Corbacho, C., J.M. Sanchez, and E. Costillo. 2003. Patterns of structural complexity and human disturbance of riparian vegetation in agricultural landscapes of a Mediterranean area. Agric. Ecosyst. Environ. 95:495–507. doi:10.1016/S0167-8809(02)00218-9

Curry, J.P. 1994. Grassland invertebrates. Chapman and Hall, London.

Dennis, P., J. Skartveit, D.I. McCracken, R.J. Pakeman, K. Beaton, A. Kunaver, and D.M. Evans. 2008. The effects of livestock grazing on foliar arthropods associated with bird diet in upland grasslands of Scotland. J. Appl. Ecol. 45:279–287. doi:10.1111/j.1365-2664.2007.01378.x

Donald, P.F., and A.D. Evans. 2006. Habitat connectivity and matrix restoration: The wider implications of agri-environment schemes. J. Appl. Ecol. 43:209–218. doi:10.1111/j.1365-2664.2006.01146.x

Downie, I.S., I. Ribera, D.I. McCracken, W.L. Wilson, G.N. Foster, A. Waterhouse, V.J. Abernethy, and K.J. Murphy. 2000. Modelling populations of *Erigone atra* and *E. dentipalpis* (Araneae: Linyphiidae) across an agricultural gradient in Scotland. Agric. Ecosyst. Environ. 80:15–28. doi:10.1016/S0167-8809(00)00130-4

EFMA. 2006. Sustaining fertile soils and productive agriculture. Eur. Fert. Manufacturers Assoc. Brussels, Belgium.

Hayes, G.F., and K.D. Holl. 2003. Cattle grazing impacts on annual forbs and vegetation composition of mesic grasslands in California. Conserv. Biol. 17:1694–1702. doi:10.1111/j.1523-1739.2003.00281.x

Haysom, K.A., D.I. McCracken, G.N. Foster, and N.W. Sotherton. 2004. Developing grassland conservation headlands: Response of carabid assemblage to different cutting regimes in a silage field. Agric. Ecosyst. Environ. 102:263–277. doi:10.1016/j.agee.2003.09.014

Hendrickx, F., J.P. Maelfait, W. Van Wingerden, O. Schweiger, M. Speelmans, S. Aviron, L. Augenstein, R. Billeter, D. Bailey, R. Bukacek, F. Burel, T. Diekotter, J. Dirksen, F. Herzog, J. Liira, M. Roubalova, V. Vandomme, and R. Bugter. 2007. How landscape structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes. J. Appl. Ecol. 44:340–351. doi:10.1111/j.1365-2664.2006.01270.x

Henle, K., A. Alard, J. Clitherow, P. Cobb, L. Firbank, T. Kull, D.I. McCracken, R. Moritz, H. Mühle, J. Niemelä, P. Nowicki, M. Rebane, D. Wascher, A. Watt, and J. Young. 2008. Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe: A review. Agric. Ecosyst. Environ. 124:60–71. doi:10.1016/j.agee.2007.09.005

Hopkins, J.J., A.J. Duncan, D.I. McCracken, S. Peel, and J.R.B. Tallowin. 2007. High value grasslands: Providing biodiversity, a clean environment and premium products. The British Grassland Society, Cirencester, UK.

Kristensen, P. 2003. EEA core set of indicators: Revised version April 2003. Technical report. Eur. Environ. Agency, Copenhagen, Denmark.

Lovell, S.T., and W.C. Sullivan. 2006. Environmental benefits of conservation buffers in the United States: Evidence, promise, and open questions. Agric. Ecosyst. Environ. 112:249–260. doi:10.1016/j.agee.2005.08.002

Marshall, E.J.P., T.M. West, and D. Kleijn. 2006. Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. Agric. Ecosyst. Environ. 113:36–44. doi:10.1016/j.agee.2005.08.036

McCracken, D.I. 2010. How effectively is water quality being managed? p. 60–69. *In* S. Skerratt, C. Hall, C. Lamprinopoulou, D. McCracken, A. Midgley, M. Price, A. Renwick, C. Revoredo, S. Thomson, F. Williams, and A. Wreford (ed.) Rural Scotland in focus. Rural Policy Centre, Scottish Agricultural College, Edinburgh, Scotland. Available at http://www.sac.ac.uk/mainrep/pdfs/rsif2010.pdf (verified 28 June 2011).

McCracken, D.I., and E.M. Bignal. 1998. Applying the results of ecological studies to land-use policies and practices. J. Appl. Ecol. 35:961–967.

McCracken, D.I., and J.R. Tallowin. 2004. Sward and structure, the interactions between farming practices and bird food resources in lowland grassland. Ibis 146:108–114. doi:10.1111/j.1474-919X.2004.00360.x

Morris, M.G. 2000. The effects of structure and its dynamics on the ecology and conservation of arthropods in British grasslands. Biol. Conserv. 95:129–142. doi:10.1016/S0006-3207(00)00028-8

Muscutt, A.D., G.L. Harris, S.W. Bailey, and D.B. Davies. 1993. Buffer zones to improve water quality: A review of their potential use in UK agriculture. Agric. Ecosyst. Environ. 45:59–77. doi:10.1016/0167-8809(93)90059-X

Niemelä, J., E. Halme, and Y. Haila. 1990. Balancing sampling effort in pitfall trapping of carabid beetles. Entomol. Fenn. 1:233–238.

Perkins, A.J., M.J. Whittingham, R.B. Bradbury, J.D. Wilson, A.J. Morris, and P.R. Barnett. 2000. Habitat characteristics affecting use of lowland agricultural grassland by birds in winter. Biol. Conserv. 95:279–294.

Pykälä, J. 2003. Effects of restoration with cattle grazing on plant species composition and richness of semi-natural grasslands. Biodivers. Conserv. 12:2211–2226. doi:10.1023/A:1024558617080

Robel, R.J., J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. J. Range Manage. 23:295–297. doi:10.2307/3896225

Robinson, R.A., J.D. Wilson, and H.Q.P. Crick. 2001. The importance of arable habitat for farmland birds in grassland landscapes. J. Appl. Ecol. 38:1059–1069. doi:10.1046/j.1365-2664.2001.00654.x

Schaich, H., M. Rudner, and W. Konold. 2010. Short-term impact of river restoration and grazing on floodplain vegetation in Luxembourg. Agric. Ecosyst. Environ. 139:142–149. doi:10.1016/j.agee.2010.07.012

Schweiger, O., J.P. Maelfait, W. van Wingerden, F. Hendrickx, R. Billeter, M. Speelmans, I. Augenstein, B. Aukema, S. Aviron, D. Bailey, R. Bukacek, F. Burel, T. Diekotter, J. Dirksen, M. Frenzel, F. Herzog, J. Liira, M. Roubalova, and R. Bugter. 2005. Quantifying the impact of environmental factors on arthropod communities in agricultural landscapes across organisational levels and spatial scales. J. Appl. Ecol. 42:1129–1139.

SEPA. 2009a. River Basin Planning. The river basin management plans for the Scotland and Tweed river basin districts. Scottish Environment Protection Agency, Stirling, Scotland. Available at http://www.sepa.org.uk/water/river_basin_planning.aspx (verified 28 June 2011).

SEPA. 2009b. Environmental objectives for rural diffuse pollution. Scottish Environment Protection Agency, Stirling, Scotland. Available at http://www.sepa.org.uk/water/river_basin_planning/diffuse_pollution_mag/environmental_objectives.aspx (verified 28 June 2011).

SEPA. 2010a. Bathing Water Quality Reports. Scotland's Bathing Waters 2009. Scottish Environment Protection Agency, Stirling, Scotland. Available at http://www.sepa.org.uk/water/water_publications/bathing_water.aspx (verified 28 June 2011).

SEPA. 2010b. The rural diffuse pollution plan for Scotland. Scottish Environment Protection Agency, Stirling, Scotland. Available at http://www.sepa.org.uk/water/river_basin_planning/diffuse_pollution_mag.aspx (verified 28 June 2011).

Siriwardena, G.M., H.Q.P. Crick, S.R. Baillie, and J.D. Wilson. 2000. Agricultural land-use and the spatial distribution of granivorous lowland farmland birds. Ecography 23:702–719. doi:10.1111/j.1600-0587.2000.tb00314.x

Smith, R.S. 1994. Effects of fertilizers on plant species composition and conservation interest of UK grassland. p. 64–73. *In* R.J. Haggar and S. Peel (ed.) Grassland Management and Nature Conservation. British Grassland Soc. Occasional Symp. 28, Leeds, UK. 23–29 Sept. 1993. British Grassland Soc., Reading, UK.

Stewart, K.E.J., N.A.D. Bourn, and J.A. Thomas. 2001. An evaluation of three quick methods commonly used to assess sward height in ecology. J. Appl. Ecol. 38:1148–1154. doi:10.1046/j.1365-2664.2001.00658.x

ter Braak, C.J.F., and P. Šmilauer. 2002. CANOCO reference manual and CanoDraw for Windows user's guide: Software for canonical community ordination (version 45). Microcomputer Power, Ithaca, NY.

Vickery, J.A., J.R.B. Tallowin, R.E. Feber, P.W. Atkinson, E.J. Asteraki, R.J. Fuller, and V.K. Brown. 2001. Changes in lowland grassland management: Implications for invertebrates and birds. J. Appl. Ecol. 38:647–664.

Weibull, A.C., J. Bengtsson, and E. Nohlgren. 2000. Diversity of butterflies in the agricultural landscape: The role of farming system and landscape heterogeneity. Ecography 23:743–750. doi:10.1111/j.1600-0587.2000.tb00317.x

Whittingham, M.J. 2007. Will agri-environment schemes deliver sufficient biodiversity gain, and if not why not? J. Appl. Ecol. 44:1–5. doi:10.1111/j.1365-2664.2006.01263.x

Wilson, J.D., A.J. Morris, B.E. Arroyo, S.C. Clark, and R.B. Bradbury. 1999. A review of the abundance and diversity of invertebrates and plant foods of granivorous birds in northern Europe in relation to agricultural change. Agric. Ecosyst. Environ. 75:13–30.

Woodcock, B.A., S.G. Potts, E. Pilgrim, A.J. Ramsay, T. Tscheulin, A. Parkinson, R.E.M. Smith, A.L. Gundrey, V.K. Brown, and J.R. Tallowin. 2007a. The potential of grass field margin management for enhancing beetle diversity in intensive livestock farms. J. Appl. Ecol. 44:60–69. doi:10.1111/j.1365-2664.2006.01258.x

Woodcock, B.A., S.G. Potts, T. Tscheulin, E.S. Pilgrim, A.J. Ramsay, J. Harrison-Cripps, V.K. Brown, and J.R. Tallowin. 2009. Responses of invertebrate trophic level, feeding guild and body size to the management of improved grassland field margins. J. Appl. Ecol. 46:920–929.

Woodcock, B.A., S.G. Potts, D.B. Westbury, A.J. Ramsay, M. Lambert, S.J. Harris, and V.K. Brown. 2007b. The importance of sward architectural complexity in structuring predatory and phytophagous invertebrate assemblages. Ecol. Entomol. 32:302–311.