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Manuscript Draft

Manuscript Number: ECOLIND-1765R1

Title: Different bioindicators measured at different spatial scales vary in their response to agricultural intensity

Article Type: Research Paper

Keywords: Key-words: agro-ecology, indicator, biodiversity, livestock farming, habitat heterogeneity, agri-environment policy

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Corresponding Author's Institution: University College Dublin

First Author: Barry John McMahon, PhD

Order of Authors: Barry John McMahon, PhD; Annette Anderson; Tim Carnus; Alvin J Helden; Mary Kelly-Quinn; Amel Maki; Helen Sheridan; Gordon Purvis

Abstract: Ecologically, potential bioindicator taxa operate at different scales within agricultural ecosystems, and thereby provide a means to investigate the influence of changing management practice on biological diversity at different scales within the agro-ecosystem. Surveys of grassland plant species at field level, parasitoid Hymenoptera at the field and farm scale, and bird populations and habitats at farm scale were carried out on 119 grass-based farms across three regions in the Republic of Ireland. In addition, habitat richness and aquatic macroinvertebrates were quantified at landscape scale. Agricultural intensity on the surveyed farms was quantified by mean farm stocking rate, calculated as livestock units per ha (LU/ha), and generalized linear mixed models used to evaluate relationships between stocking rate and the incidence of chosen bioindicator groups. Field scale bioindicators (plant species richness and parasitoid taxon richness and abundance) were negatively associated with mean farm stocking rate. Over much of its observed range, mean farm stocking rate was positively associated with total bird species richness and abundance, and the species richness and abundance of Farmland Bird Indicator species recorded in the winter season. However, these relationships were quadratic, and above a relatively high upper limit of 2.5-3.5 LU/ ha, further increase in farm stocking rate had a negative influence. Results demonstrate that different bioindictors measured at different spatial scales vary in their response to agricultural intensity. The lack of a consistent bioindicator response to farm stocking rate suggests that within predominantly farmed regions, maximising biodiversity requires a careful targeting and monitoring with bioindicator taxa that are informative of influences at relevant operational scales. The insights provided may then be much more informative for the design and implementation of agri-environment measures that maximise biodiversity within farmed landscapes.

Response to Reviewers: Dear Prof. Dr. Felix Müller

Thank you for your email of the 27nd November and the reviewers comments on our submission. In revising the MS as you suggested, we have tried to attend to the important and extremely useful comments. In response individual comments of the reviewers are listed below.

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3. Does the paper support the progress in indicator development or does it provide interesting and new indicator applications?
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10. Are the statistical methods used correctly and adequate?
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11. Are the results clearly presented?
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Dear Author/Authors,
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For the "Conclusions" section, it would be great to consider the possibility of adding some recommendations for further researches and for indicators for policy agencies;

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Barry J. McMahon[a], Annette Anderson[a], Tim Carnus[a], Alvin J. Helden[a,b], Mary Kelly-Quinn[c], Amel Maki[c], Helen Sheridan[a,]* and Gordon Purvis

[a]UCD School of Agriculture and Food Science University College Dublin, Belfield, Dublin 4, Ireland

[b]Animal and Environmental Research Group, Department of Life Sciences, Anglia Ruskin University, East Road, Cambridge, UK

[c]UCD School of Biology and Environmental Science, University College Dublin Belfield, Dublin 4, Ireland

*Corresponding author: Helen Sheridan, UCD School of Agriculture and Food Science, University College Dublin, Belfield, Dublin 4, Ireland (Tel: +353 1716 7010; fax: +353 1716 1102; e-mail: helen.sheridan@ucd.ie)

ABSTRACT

Ecologically, potential bioindicator taxa operate at different scales within agricultural ecosystems, and thereby provide a means to investigate the influence of changing management practice on biological diversity at different scales within the agro-ecosystem. Surveys of grassland plant species at field level, parasitoid Hymenoptera at the field and farm scale, and bird populations and habitats at farm scale were carried out on 119 grass-based farms across three regions in the Republic of Ireland. In addition, habitat richness and aquatic macroinvertebrates were quantified at landscape scale. Agricultural intensity on the surveyed farms was quantified by mean farm stocking rate, calculated as livestock units per ha (LU/ha), and generalized linear mixed models used to evaluate relationships between stocking rate and the incidence of chosen bioindicator groups. Field scale bioindicators (plant species richness and parasitoid taxon richness and abundance) were negatively associated with mean farm stocking rate. Over much of its observed range, mean farm stocking rate was positively associated with total bird species richness and abundance, and the species richness and abundance of Farmland Bird Indicator species recorded in the winter season. However, these relationships were quadratic, and above a relatively high upper limit of 2.5-3.5 LU/ ha, further increase in farm stocking rate had a negative influence. Results demonstrate that different bioindictors measured at different spatial scales vary in their response to agricultural intensity. The lack of a consistent bioindicator response to farm stocking rate suggests that within predominantly farmed regions, maximising biodiversity requires a careful targeting and monitoring with bioindicator taxa that are informative of influences at relevant operational scales. The insights provided may then be much more informative for the design and implementation of agri-environment measures that maximise biodiversity within farmed landscapes.

Key-words: agro-ecology, indicator, biodiversity, livestock farming, habitat heterogeneity, agri-environment policy

1. Introduction

Agricultural intensification has been held responsible for a marked reduction in biodiversity across north-west Europe in recent decades (Donald et al., 2001; Benton et al., 2003). The process of intensification brings about multiple coincident changes, which in livestock farming involve much more than a simple increase in stocking rates and greater use of nutrient inputs. Other significant effects include an up-scaling of the size of farms and individual production units (fields) with coincident loss of non-cropped habitats such as permanent field boundaries/hedgerows, and an increased specialisation of the farming system. The result is a greater homogenisation of the landscape within farming regions, with reduced habitat diversity and spatial heterogeneity. This has been labelled as one of the principle reasons for declining farmland biodiversity over recent decades (McLaughlin and Mineau, 1995; Duelli, 1997; Robinson and Sutherland, 2002; Benton et al., 2003; Hoffmann and Greef, 2003). Conversely, the opposite of agricultural intensification, i.e. abandonment of traditional land management practice, poses a potentially equal threat to biodiversity within economically marginal farming regions (Henle et al., 2008) and all manner of agricultural land within central and eastern Europe. Such changes in farming have led to an increasingly important practical debate, as to how agri-environment policy can best mitigate the detrimental effects of changing management practice.

In seeking to better understand the ecological effects of changes in farming practice, the identification and use of indicators of biodiversity status within agro- ecosystems has been the focus of much debate and research over the last decade (McGeoch, 1998; Büchs, 2003, Purvis et al. 2009a). There can be no single bioindicator for all aspects of biodiversity in all contexts (McGeoch, 1998). In order for indicators to be used to their fullest advantage, it is necessary to understand the ecological relationships between the chosen indicator group(s) and wider community structure, as well as the particular ecological influences they reflect (Paoletti, 1999). Plants, parasitoid Hymenoptera (hereafter referred to as parasitoids) and birds have all been utilised in recent studies as potential biodindicators within agricultural landscapes (Scholefield et al., 2010; McMahon et al., 2010a; Anderson et al., 2011), and aquatic macroinvertebrates have an established role as bioindicators, for example of water quality (Armitage et al., 1983; Metcalfe, 1989). The influence of scale, relating to the mobility, ecology and processes that influence the chosen group, is increasingly recognised as potentially relevant to indicator utility, and the insights and information they provide (Duelli, 1997).

In practice, agri-environment schemes (AES) are largely targeted at influencing the management of individual farmers (Purvis et al., 2009b), and so policy has tended to focus on measures targeted at the farm scale. Despite their questionable effectiveness (Kleijn and Sutherland, 2003), a very significant investment has been made in these schemes, most of which make an implicit assumption that all aspects of agricultural intensification at the farm level are always detrimental to farmland biodiversity. However, it is clear that the effects of changing farming practice operate at a range of scales, from field to farm and landscape levels (Gabriel et al., 2010). A greater understanding of how different indicator groups might be used to document and interpret the relative importance of such effects would potentially benefit the design of more effective policy measures. There is little consensus as to the relative importance of the different elements of changing farming practice on farmland biodiversity, or knowledge of how bioindicator groups that are indicative of effects at different scales can be used to evaluate such effects within the agro-ecosystem.

Agriculturally managed grasslands represent one of the most important forms of land use, accounting for almost 68% of total farmland at a global scale (Anon., 2009a). Within the Republic of Ireland (ROI), approximately 6,900,000 ha of land are devoted to farming, which represents 62% of the total land area. Approximately 80% of this agricultural land is devoted to grass-based livestock farming, including intensively grazed pasture and grass forage production (DAFF, 2009). The intensification of grassland management in Irish farming, especially through changes in reseeding and the frequency of new sward establishment, grazing and forage conservation systems and nutrient inputs, has mirrored the intensification of agriculture generally across much of Europe, which has resulted in an associated loss of biodiversity, including botanical biodiversity (Kleijn et al., 2009). However, despite lowland grassland farming being the most widespread form of land use in the ROI, it has until relatively recently remained one of least studied ecosystems. A number of bioindicator groups have been proposed for agricultural ecosystems such as bees and butterflies (Santorumn and Breen 2005; Rundlöf et al 2008). In this study, systematic use is made of data collected to examine the relationships between agricultural intensity quantified by farm stocking rate, and bioindicator groups chosen to reflect processes and influences at different scales ranging from individual fields to the farm and landscape level. However, the selection of the bioindicators groups in this study was principally informed by previous research on lowland agricultural grasslands within the ROI (Purvis et al, 2009a), but also to reflect the different scales at which farming may influence biodiversity. To our knowledge this is the first attempt to integrate information regarding such a wide range of bioindicators that operate at a range of scales within an agricultural landscape.

2. Methods

2.1. Site Selection

Grass-based farms were selected in three separate geographical regions, in counties Sligo/Leitrim (north-west), Offaly/Laois (central) and Cork (south) of the ROI. The selected regions represent a farming intensity gradient, reflected by a preponderance of extensive non-dairy farming in Sligo/Leitrim and intensive dairy farming in Cork, with a mixed farming economy of non-dairy and dairying in Offaly/Laois (Lafferty et al., 1999).

In March 2007 and 2008, five 10 x 10 km squares (henceforth referred to as 10km squares) were randomly selected from the Ordinance Survey Ireland (OSI) map within each study region, and from within each of these main sampling squares, an individual farm was surveyed at the centre of each of the four central 1km squares. Only 10 km squares under 250 m in elevation, and with at least 70% agricultural land cover were included in the selection process. In total, sixty farms were surveyed in 2007-08 (3 regions x 5 main squares x 4 farms), and 59 farms in 2008-09 (after failure to find a fourth co-operative farmer in a square selected within the Cork region).

2.2. Plant data

A permanent internal field boundary was chosen on each of the farms which was adjacent to a permanent grass sward that had not been reseeded for at least 5-years, and that faced south-west or as close to a south-west orientation as was feasible, was selected. For the purpose of this study, a field boundary was defined as a permanent hedgerow following McMahon et al., (2010b). Botanical diversity was assessed within two 1x1 m^2 quadrats along four transects within each field. Transects were located perpendicular to the field boundary, at distances of at least 10 m from field boundary intersection points, with a minimum of 10m between each. Quadrats were positioned at distances of 0.5-1.5 m ('Field margin') and approximately 20 m from the field boundary ('Field') along each transect. All specimens rooted within the quadrat area were identified to species level (Stace, 1997), except in situations when frequent hybridisation is known to occur, in which case they were identified to genus level e.g. *Agrostis* sp. Species abundance was recorded according to the Braun-Blanquet Scale (Kent and Coker, 1992).

2.3. Parasitoid data

Parasitoids collected from associated field swards were sampled using a Vortis Insect Suction Sampler (Burkard Manufacturing Co Ltd, Rickmansworth, Hertfordshire, UK) (Arnold, 1994). Ten samples, each consisting of an aggregate of 6 randomly selected sampling spots, individually sampled for ten seconds, were collected from the centre of the randomly-chosen grassland field on each farm (i.e. no closer than 20 m from the field edge). The total area sampled per field was therefore 1.2 m^2.

Yellow pan traps with a window interceptor (Calabuig, 2000) with water and detergent to reduce surface tension were used to collect mobile flying parasitoid populations (Gibb and Oseto, 2005). Three traps, sited on posts approximately 1m above ground level, were located within 0.5-1.5 m of the monitored (south-west facing) field boundary on each farm, at intervals of at least 10 m. Resulting pan trap catches were collected after 48 hours and transferred to storage in 70% ethanol. Parasitoids were identified to at least genus, and where possible species level using the literature cited by Anderson et al., (2008).

2.4. Bird data

Each farm was surveyed once in the breeding season (April-June) and once in the winter season (December-February). The same surveyor (BJMcM) carried out all surveys according to a standardised protocol. During each survey, field boundaries across the farm were walked at a distance of approximately 1.5 m from the field edge. The speed of walking depended on the numbers of birds present; however, because of the open nature of farmland habitats the recommended average speed of 2 km per hour was maintained where possible (Bibby *et al.* 2000). The route of each survey was consistent within each site in the breeding and winter season. Bird species presence and abundance was recorded using both visual (10 x 42 binoculars) and aural methods. In addition, because some species are known to avoid or prefer field boundaries, pre-determined transects included walking across larger fields (Chamberlain et al., 1999; Bibby et al., 2000). During the breeding season, surveys were carried out between 07.00 and 12.00 and between 10.00 and 15.00 in the winter season in order to standardise the time of day each survey was carried out within each season. The mean duration (± SD) of surveys in the winter season was 61 ± 13 minutes and 67 ± 18 minutes in the breeding season. As extreme weather affects bird activity and observer accuracy (Bibby et al., 2000), no surveys were carried during periods of persistent heavy rain, or wind speeds greater than Beaufort scale 4. The number, abundance and location of bird species were recorded directly onto site maps, including raptors seen hunting over fields and field boundaries. Other species seen flying overhead, but not interacting with fields or field boundaries, were not recorded (Perkins et al., 2000).

2.5. Habitat data

Habitat data were collected at the scale of monitored farms, and at the scale of the 16 central 1 x 1 km squares within each main (10 x 10 km) sampling square (effectively the central 4 x 4 block of 1 km squares within each main 10km survey square, including the central four in which surveyed farms were located). Farm habitat surveys were undertaken by walking the principal holding managed by each farmer and recording the type and extent of all habitats on farm maps. Classification of habitats generally followed the designations of Fossitt (2000). As this standard reference to Irish terrestrial habitats makes little distinction between agriculturally managed grasslands, additional habitat categories based on sward botanical composition were also recorded as detailed by Sheridan et al., (2011). All recorded farm habitats, including the number and length of permanent farm boundaries were digitised onto Ordnance Survey Ireland (OSI) orthophotographs (2004) using ArcGIS software. The total area occupied by field boundaries was quantified. In addition, the total area of semi-natural habitats was quantified as farm area excluding agriculturally productive areas and farm buildings.

Farm scale habitat survey information was then used as ground-truth data to classify habitats within the approximately 4 x 4 km surrounding the farms. Unsupervised classification of landscape scale habitats was undertaken using Spot satellite imagery and MultiSpec and ArcGIS software. The extent of all habitats recorded both at farm and landscape scales was then standardised as estimates per ha at farm scale and per km^2 at landscape scale, respectively.

2.6. Aquatic macroinvertebrate data

Standard multi-habitat, 'kick sampling' of aquatic macroinvertebrate was undertaken in 67 watercourses across the 30 surveyed 10 km squares. Each watercourse was sampled in spring 2007 (a total of 36 watercourses) and spring 2009 (31) and in autumn 2007 (33) and autumn 2008 (31). Time was spent proportionately during the sampling process in the riffles, pools and margins as per the percentage occurrence of each habitat at the site (covering approx. 50m) (Wright 1995). Habitats contributing less than 5% of the stable habitat in the reach were not sampled (Barbour et al., 1997). Sampling was initiated downstream of the reach and proceeded upstream. Samples were preserved in 70% IMS, and sorted and identified in the laboratory. The identifications were made to the lowest taxonomic unit possible species/genus for the Plecoptera, Ephemeroptera, Trichoptera, Coleoptera, Mollusca and Hirudinea and to family/sub-family level for most of the other groups. From these data, three internationally recognised biological watercourse quality indicators were compiled for each sample taken. These comprised the Biological Monitoring Working Party (BMWP) Index, the Average Score Per Taxon (ASPT) Index and the Ephemeroptera-Plecoptera-Trichoptera (EPT) Index (Armitage et al., 1983; Lenat, 1988).

2.7. Data analysis

Animal stocking rate, calculated as standardised livestock units per ha (LU/ha), was calculated as a measure of overall agricultural intensity on the surveyed farms, following the methodology of the Irish National Farm Survey (Anon., 2009b). Although the majority of livestock were cattle (beef, dairy and suckler) some farms also stocked sheep.

In recognition of likely potential correlations between observations made within each 10 km square, the relationship between stocking rate and the chosen indicators of farmland biodiversity (likely to be informative regarding management influences operating at different scales - Table 1), was assessed using Generalized Linear Mixed Models (GLMM). Poisson distribution was specified when residual deviance approximated to the number of degrees of freedom. If overdispersion was detected in a response variable, an observation/farm level random effect was also included in the model (full model deviance/residual df > 2). When significant, region and year were included as blocking factors in all models, and centred ordinal date on which the farms were sampled was included as a primary covariate. The effect of stocking rate (linear or quadratic) was assessed by likelihood ratio tests. As the primary objective was to establish the existence (or not) of significant relationships between indicator statistics and our chosen measure of farming intensity, we refrain in the current paper from any further analysis to elucidate potential underlying mechanisms. One exception to this, however, was the further exploration of relationships between habitat statistics and bird population statistics (response variables), again using GLMMs as described above. For all models used to analyse bird data, centred and log-transformed survey duration (minutes) was included as an offset variable, and farm area was also included as a covariate. Landscape habitats and aquatic macroinvertebrate data analyses was carried at the level of the 10km square and stocking rate was averaged across the four surveyed farms within each 10km square.

In addition, the relationship between the bird response variables for the breeding and winter season was tested with farm habitat richness, farm field boundary density and landscape habitat richness. All analyses were performed in R 2.12 (R Development Core Team, 2010).

Insert Table 1.

3. Results

No consistent response to stocking rate was found in modelled indicators. Responses varied from significantly positive (quadratic) to negative (linear), whilst a number of potential indicators had no significant relationship with our chosen measure of farming intensity.

3.1. Plant data

A total of 174 plant species was recorded in the centre and margins of surveyed fields (Table S1). There was a significant negative relationship between stocking rate and sward species richness at field centres, but no such relationship was found between stocking rate and plant species richness at field margins (Table 2, Fig. 1a, b).

3.2. Parasitoid data

A total of 9,343 parasitoids, representing 228 indentified taxa were recorded in Vortis suction samples from field centres. Parasitoid taxon richness was negatively influenced by increased stocking rate (Table 2, Fig. 1c). There was a weak negative relationship between parasitoid abundance and increased stocking rate (Table 2, Fig. 1d). A total of 5,984 parasitoid wasps, representing 487 taxa of parasitoids were recorded in pans traps catches. No significant relationship was found between stocking rate and either the abundance or taxon richness of these catches (Table 2).

3.3. Bird data

A total of 4,055 individual birds, representing 50 species were recorded during the breeding season, and a total of 11,892 individuals, representing 55 species were recorded in the winter season. A full listing of species recorded in each season is presented in the Table S2. There was a positive relationship between stocking rate and total bird species richness and abundance recorded in the winter season, and also on the species richness and abundance of Farmland Bird Indicator species (Gregory et al., 2004) recorded at this time of year (Table 2, Fig. 1e, f). These relationships were quadratic, with winter bird statistics increasing positively up to an optimal upper stocking rate, thereafter declining. This optimal upper point ranged between approximately 2.5-3.5 LU/ha (Table 2.). The quadratic relationship between winter bird abundance and stocking rate was weak (Table 2.) No significant relationships were observed between stocking rate and birds recorded in the breeding season.

There was a significant negative relationship between farm habitat richness and winter bird abundance (χ^2= 4.00, P= 0.046) during the winter season. In addition, there was negative relationship between landscape habitats and species richness of Farmland Bird Indicators (χ^2= 5.70, P= 0.017) during the winter season.

3.4. Habitat data

There was a significant relationship between stocking rate and total area of semi-natural habitat (Table, Fig.1h). No significant relationship was found between stocking rate and the number of habitats recorded in farm surveys, or the calculated density of field boundaries per farm (Table 2, Fig. 1h). No significant relationship was found between mean farm stocking rate per 10 km square and the number of habitats recorded in the quantification of habitat richness at the landscape (16 x 1 km square) level (Table 2, Fig. 1i).

3.5. Aquatic macroinvertebrates data

A total of 586,421 invertebrate individuals were identified to species/genus/family/sub-family (total = approximately 300 species; this figure is approximate because some taxa could not be positively identified). There was no significant relationship between mean farm stocking rate per 10 km square and BMWP Index, the ASPT Index and the EPT Index (Table 2).

Insert Fig. 1.

4. Discussion

The results of this study demonstrate that different bioindictors measured at different spatial scales vary in their response to agricultural intensity. Increased nutrient input levels can influence both sward plant and arthropod communities in grasslands, with a generally negative effect on species richness (Haddad et al., 2000; Klimek et al., 2007; Prestige, 1982; Zechmeister et al., 2003). In a recent study of 117 European grasslands, Klimek et al., (2007) concluded that a reduction in both nitrogenous fertiliser input and stocking rates might be important in conserving biodiversity within agricultural grasslands. Increased grassland management intensity has generally also been found to decrease associated arthropod biodiversity, and practices such as increased fertiliser use, grazing, cutting, ploughing and reseeding are likely to reduce biological diversity (Plantureux et al., 2005). In particular, the heavy grazing associated with higher stocking rates, produces short swards that reduce foraging opportunities and structural habitat diversity within swards for many invertebrates (Morris, 2000), whilst low stocking rates can favour groups like spiders, whose incidence is strongly dependant on vegetation structure (Plantureux et al., 2005). In addition, the findings of our study may not be entirely applicable beyond grassland ecosystems e.g. in arable productions systems, in the nature of how specific taxa respond to intensity. However, the variation in how different bioindictors measured at different spatial scales respond to agricultural intensity may very well be.

It is therefore not surprising that our data revealed a significantly negative influence of stocking rate on sward species richness in the centre of surveyed fields and the abundance and diversity of parasitoid wasps within the sward; the latter group being good indicators of taxon richness of arthropod populations within agricultural grasslands (Anderson et al., 2011). It is noteworthy, however, that neither botanical diversity at the margins of fields, or the abundance and diversity of more mobile flying parasitoid populations caught in window pan traps close to the boundary of surveyed fields showed such an effect.

In marked contrast, all observed winter statistics for bird populations, including the abundance and species richness of Farmland Indicator species, showed a quadratic relationship, and positive influence of increased stocking rate up to relatively high levels of between 2.5-3.0 LU/ha. It is important to note that very few surveyed farms had stocking rates in excess of this level, which is probably close to the maximum achievable under Irish conditions within the constraints imposed by legislation such the Nitrates Directive (DAFF, 2004). No such stocking rate influence was found for bird populations in the breeding season, and perhaps tellingly, no significant relationships were found between stocking rate and any observed measure of habitat richness at landscape level.

The negative relationship between farm habitat richness and winter bird abundance and landscape habitat richness and species richness of Farmland Bird Indicators in the winter was not expected. However, these relationships could possibly be explained by the fact that food resources are more important during the winter season and a more heterogeneous landscape may actually reduce the availability of such resources. Birds may acquire greater food resources provided by large areas of improved grasslands, particularly invertebrate feeding species. Findings from the bird habitat models indicate that a more extensive investigation of our dataset is required to full understand the relationship between agricultural habitats, farming intensity and farmland birds.

The existence of a positive stocking rate influence, and by inference a positive influence of overall management intensity within managed grassland fields on winter bird populations, is counter-intuitive and contradicts any assumption that grassland management intensity has a negative impact on all aspects of farmland biodiversity. Perhaps our results can best be explained in light of previous work suggesting that food availability (trophic energy) is a key factor in determining bird species diversity (Haberl et al., 2005), and that production intensity can have a positive influence on some specialist farmland bird species (Donald et al., 2006). Indeed, previous studies have shown that some winter bird populations occur in greater numbers on intensively managed fields (Atkinson et al., 2005), in which soil invertebrates, especially earthworms can be significantly more abundant (if not more diverse) under conditions of greater nutrient input levels (Curry et al., 2008).

At the landscape scale, water quality has been linked to catchment characteristics and intensity of agricultural activities (e.g., Genito et al., 2002; Donohue et al., 2006; Rothwell et al., 2010). The number of sensitive taxa, as represented by indices such as EPT percentage composition, is known to reflect anthropogenic inputs (Resh and Jackson, 1993). In a previous study by Baars and Kelly-Quinn (2005) differences between intensively agricultural and reference sites were highlighted using the metrics applied in the present study. The lack of effect detected in this larger study is not totally unexpected as it is likely that the stocking rate derived from the average across the four surveyed farms within each 10km squares was not truly representative of the watershed of the study sites.

A careful selection of appropriate indicators is needed to understand the underlying relationships between changing farming practice and biodiversity within any particular farming context. Within individual grassland fields, sward and closely associated arthropod diversity are negatively impacted by increasing stocking rate, and by inference, increasing intensity of grassland management. However, these effects do not necessarily extend to field margins, or to more mobile taxa dependent on other resources within the farmed landscape. In particular, our data suggest that provided other necessary resources such as the extent of suitable non-cropped habitats are retained, including hedgerows and other permanent field boundaries, more mobile populations within the farmed landscape may actually benefit from within-field intensity.

This relationship revealed by our data implies that enhancement of sward botanical species richness and supporting the diversity of associated invertebrate populations, is not necessarily a prerequisite to optimising the environment for farmland birds (Atkinson et al., 2005). As a part of the wider heterogeneity of the farmed landscape, intensive grassland management may play a positive role and represent an opportunity rather than a threat for taxa that utilise the agri-environment at the wider farm and landscape scale (Haberl et al., 2005).

These findings emphasise the importance of the scale, as well as the intensity of production practices (Fuller et al., 2005; Gabriel et al., 2010). In Ireland, increased intensity of grassland management has not yet resulted in a parallel process of up-scaling production units at the farm and landscape level. The density of non-cropped habitats is far greater in Ireland, relative to other farming areas in Europe (Sheridan et al., 2011). In this regard, pastoral farming in Ireland may be quite atypical, in that increased production intensity elsewhere is almost invariably accompanied by a substantial loss of non-cropped habitats (Benton et al., 2003), and most especially traditional field boundaries, within the farmed landscape.

There is widespread acceptance that the enhancement of ecological heterogeneity at multiple spatial and temporal scales is key to reversing the decline in biodiversity within agricultural ecosystems (Benton et al., 2003; McMahon et al., 2008). Heterogeneity of farmland habitats and farming systems (including production intensity), may all be important factors in determining overall biodiversity. If so, effective agri-environment policy requires the implementation of appropriate measures at multiple spatial scales, in order to maximise the delivery of a broad spectrum of ecosystem services. In Ireland, as in the majority of EU States, a single nationwide implementation of agri-environmental policy under the Rural Development

Regulation (EC) 1698/2005 (as amended by Council Regulation (EC) 74/2009), targets a broad range of agri-environment objectives, including the protection of biodiversity (Purvis et al., 2009b). It does so, by placing a particular emphasis on limiting within-field production intensity. Our findings suggest that the continued maintenance of the unusually high relative incidence of non-cropped habitats, including traditional field boundaries within the Irish farmed landscape, may along with the continued economic viability of relatively small individual farms, be the greatest priority for Irish AE-policy measures. This may be particularly so, in the light of a recent Irish Foresight Study (DAFF, 2010), that envisions an opportunity to realise a 50% increase in output from the Irish dairy sector. Such an increase is unlikely to be achievable through further increase in within-field production intensity (especially under the constraints imposed by the Nitrates Directive – DAFF, 2004), but is much more likely to be targeted through an up-scaling of production units and land use.

5. Conclusions

The measurement of biological taxa at a single scale in response to land-use activities oversimplifies ecosystems and can lead to biased results in relation to the effect on overall biodiversity. Appropriate measurement of multiple taxa at multiple scales provides critical information needed to understand the structure, function and dynamics of the complex ecosystems which reflect the real world (Jelinski et al., 1996). Both research and policy should reflect this. Information obtained from a multi-scale assessment of land use and habitat mosaics are required to inform appropriate plans to create connectivity and a matrix which can facilitate the maintenance or enhancement of regional (Zaccarelli et al., 2008). Agricultural landscapes are in a constant state of flux in response to changing societal needs. Strategies to maximise biodiversity within agricultural ecosystems need to be implemented, not at a national scale, but at a much more focused local scale that recognises regional variation and circumstance (Whittingham et al., 2007). Focus on the wider maintenance of biodiversity needs to shift to multiple scales, possibly even beyond the targeting of management practice at total farm level (Rundolf et al., 2008; Gabriel et al., 2010). The application of this principle is probably relevant to all conservation, not just that within agricultural ecosystems (Gabriel et al., 2010). Our data clearly indicate a need to utilise the information that can be provided by indicators reflective of effects at different scales, ranging from within-field, to farm and landscape levels. Only by understanding the complex ecological influences of changing farm practice at different scales, can the implementation of agri-environment policy be made maximally effective.

Acknowledgements

Funding was provided by the Irish Dept. of Agriculture Fisheries and Food Research Stimulus Fund 2006 (06 0376). The co-operation of farmers who provided sites was greatly appreciated. Thanks to Dan Chamberlain and Josef Settle for comments on the manuscript and to Dirk Miksche for his assistance with collection, sorting and identification of samples.

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Table 1. Likely bioindicators of farm management effects at different scales that were evaluated as response variables reflecting the influence of farming intensity measured as farm stocking rate (LU/ha), using Generalized Linear Mixed Models.

Indicator	Potential Scale of Influence
Field plant species richness	Field
Field margin plant species richness	Field
Field parasitoid taxon richness	Field
Field parasitoid abundance	Field
Pan trap parasitoid taxon richness	Field/Farm
Pan trap parasitoid abundance	Field/Farm
Bird species richness in the winter and breeding season	Farm/landscape
Bird species abundance in the winter and breeding season	Farm/landscape
Habitats richness (farms)	Farm
Field Boundaries (% per total farm area)	Farm
Semi-natural habitats (ha)	Farm
Habitats richness (4 x 1km squares)	Landscape
Aquatic macro invertebrates indices (10km square)	Landscape

Table 2. Summary of likelihood ratio tests (χ^2) for the effect of stocking rate (linear) and stocking rate^2 (quadratic) on selected indicators. In the case of significant quadratic model effects, optimal stocking rate are also provided.

Response Variable	**Stocking Rate Effect**		**Stocking Rate^2 Effect**		**Optimal stocking rate (LU/ha)**
	χ^2	*P value*	χ^2	*P value*	
Field plant species richness	5.55	0.019	ns	ns	-
Field margin plant species richness	ns	ns	ns	ns	-
Field parasitoid taxon richness	5.15	0.023	ns	ns	-
Field parasitoid abundance	3.36	0.067	ns	ns	-
Pan parasitoid taxon richness	ns	ns	ns	ns	-
Pan parasitoid abundance	ns	ns	ns	ns	-
Winter bird species richness	4.56	0.033	4.54	0.033	2.510
Winter bird abundance	15.85	<0.001	2.78	0.095	3.534
Winter Farmland Bird Indicator Species richness	8.55	0.003	4.78	0.029	2.998
Winter Farmland Bird Indicator Species abundance	16.23	<0.001	5.58	0.018	2.785
Breeding birds species richness	ns	ns	ns	ns	-
Breeding bird abundance	ns	ns	ns	ns	-
Breeding Farmland Bird Indicators species richness	ns	ns	ns	ns	-
Breeding Farmland Bird Indicators abundance	ns	ns	ns	ns	-
Farm habitat richness	ns	ns	ns	ns	-
Semi-natural habitats ((% per total farm area)	4.29	0.038	ns	ns	-
Farm field boundary density (% per total farm area)	ns	ns	ns	ns	-
Habitats richness(4 x 1km squares)	ns	ns	ns	ns	-
BMWP Index	ns	ns	ns	ns	-
ASPT Index	ns	ns	ns	ns	-
EPT Index	ns	ns	ns	ns	-

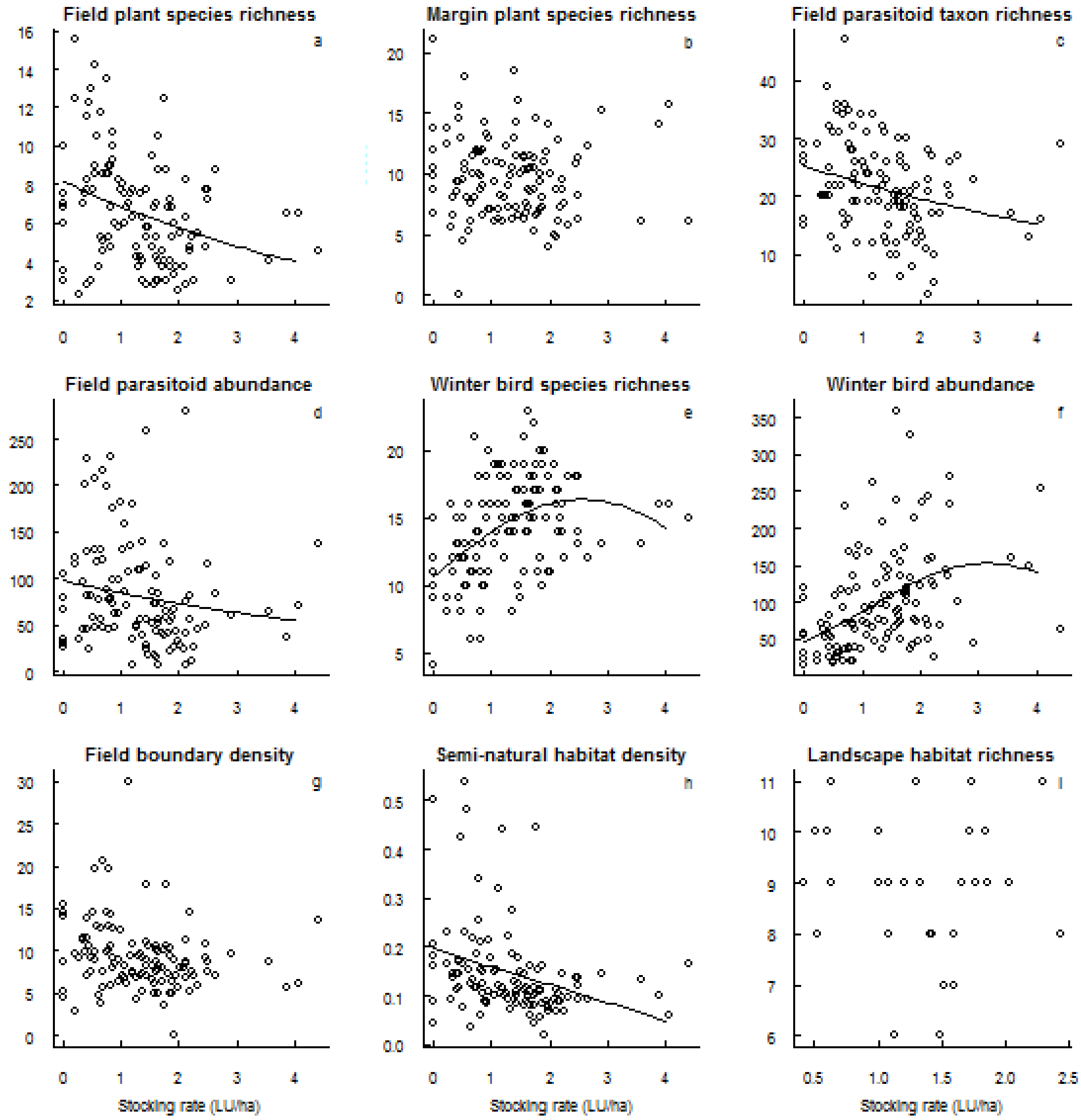

Fig. 1. Relationships between farm stocking rate and a selection of farmland biodiversity indicators. Fitted lines represent model predictions for the significant terms presented in Table 2: a) Field plant species richness, b) Field margin plant species richness, c) Field parasitoid taxon richness, d) Field parasitoid abundance, e) Winter bird species richness, f) Winter bird abundance, g) Density of farm field boundaries, h) Semi-natural habitats i) Landscape habitats richness. Note, stocking rate is expressed at the farm level in all models, except (i) for which stocking rate was averaged across the four surveyed farms within each 10km square. All response variables are counts apart from (h) and (i) which is measured as a percentage of total farm area.

Figure(s)
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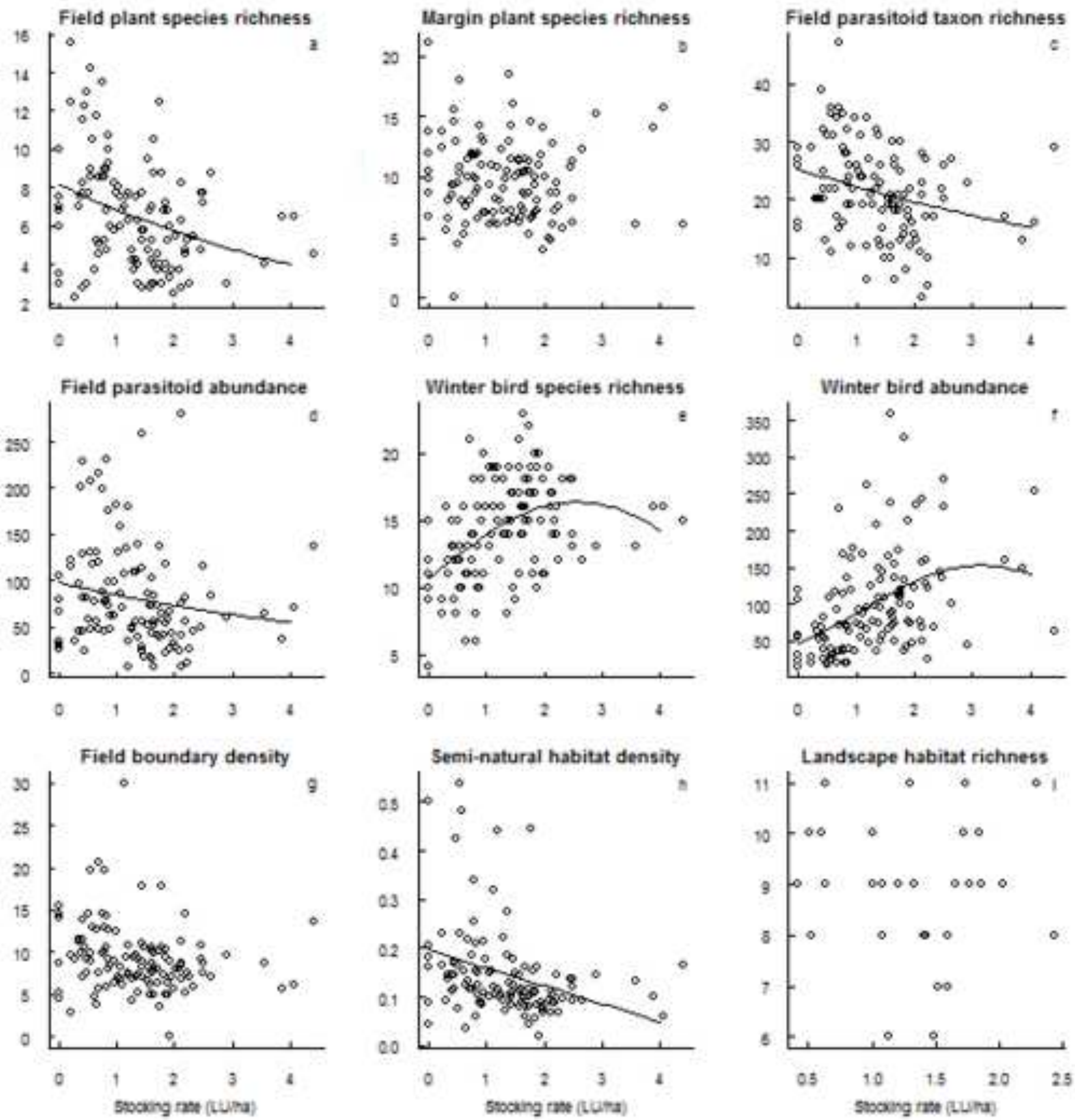

Table S1. The plant species recorded during the study

Common Name	Scientific name
Sycamore	*Acer pseudoplatanus*
Yarrow	*Achillea millefolium*
Ground elder	*Aegopodium podagraria*
Bent grass	*Agrostis sp*
Bugle	*Ajuga reptans*
Marsh foxtail	*Alopecurus geniculatus*
Foxtail	*Alopecurus pratensis*
Wild angelica	*Angelica sylvestris*
Sweet vernal	*Anthoxanthum odoratum*
Cow parsley	*Anthriscus sylvestris*
Lesser burdock	*Arctium minus*
False oat	*Arrhenatherum elatius*
Harts tongue	*Asplenium scolopendrium*
Daisy	*Bellis perennis*
Downy Birch	*Betula pubescens*
Rape	*Brassica napus*
Quaking grass	*Brizia media*
Buckler fern	*Buckler fern*
Heather	*Calluna vulgaris*
Hedge bindweed	*Calystegia sepium*
Shephards purse	*Capsella bursa pastoris*
Wavy bittercress	*Cardamine flexuosa*
Sedge species	*Carex* species
Common yellow sedge	*Carex demissa*
Glaucous sedge	*Carex flacca*
Black sedge	*Carex nigra*
Carnation sedge	*Carex panicea*
Black knapweed	*Centaurea nigra*
Common mouse ear	*Cerastium fontanum*
Rosebay willowherb	*Chamerion angustifolium*
Goosefoot	*Chenopodium album*
Enchanters nightshade	*Circaea lutetiana*
Creeping thistle	*Cirsium arvense*
Marsh thistle	*Cirsium palustre*
Spear thistle	*Cirsium vulgare*
Hazel	*Corylus avellana*
Whitethorn	*Crataegus monogyna*
Smooth hawksbeard	*Crepis capillaris*
Beaked hawksbeard	*Crepis vesicaria*
Crested dogs tail	*Cynosurus cristatus*
Cocksfoot	*Dactylis glomerata*
Common spotted orchid	*Dactylorhiza fuchsii*
Foxglove	*Digitalis purpurea*
Common sundew	*Drosera rotundifolia*
Male fern	*Dryopteris filix mas*
Common couch	*Elymus repens*

Broad-leaved willowherb	*Epilobium montanum*
Great willowherb	*Epilobium hirsutum*
Horsetail	*Equisetum arvense*
Common cotton grass	*Eriophorum angustifolium*
Bell heather	*Erica cinerea*
Sheeps fescue	*Festuca ovina*
Red fescue	*Festuca rubra*
Meadowsweet	*Fillipendula ulmaria*
Dropwort	*Fillipendula vulgaris*
Common cleavers	*Galium aparine*
Marsh bedstraw	*Galium palustre*
Heath bedstraw	*Galium saxatile*
Ladys Bedstraw	*Galium verum*
Cut leaved Cranesbill	*Geranium dissectum*
Herb robert	*Geranium robertianum*
Herb bennet / wood avens	*Geum urbanum*
Ground ivy	*Glechoma hederacea*
Flote grass	*Glyceria fluitans*
Ivy	*Hedera helix*
Hogweed	*Heracleum sphondylium*
Fog	*Holcus lanatus*
Creeping softgrass	*Holcus mollis*
Slender St Johns wort	*Hypericum pulchrum*
Holly	*Ilex aquifolium*
Yellow iris	*Iris pseudacorus*
Deer Grass	*Trichophorum cespitosum*
Sharp-flowered rush	*Juncus acutiflorus*
Juncus bufonius	*Juncus bufonius*
Soft rush	*Juncus effusus*
Hard rush	*Juncus inflexus*
Red deadnettle	*Lamium purpureum*
Nipplewort	*Lapsana communis*
Meadow vetchling	*Lathyrus pratensis*
Ox-eye daisy	*Leucanthemum vulgare*
Wild privet	*Ligustrum vulgare*
Perennial rye grass	*Lolium perenne*
Italian ryegrass	*Lolium multiflorium*
Birdsfoot trefoil	*Lotus corniculatus*
Wood rush	*Luzula campestris*
Ragged robin	*Lychnis flos cuculi*
Yellow pimpernel	*Lysimachia nemorum*
Scented mayweed	*Matricaria recutita*
Black meddick	*Medicago lupulina*
Purple moor-grass	*Molinea caerulea*
Field forgetmenot	*Myosotis arvensis*
Bog asphodel	*Narthecium ossifragum*
Adderstongue	*Ophioglossum vulgatum*
Lousewort	*Pedicularis sylvatica*

Redshank	*Persicaria maculosa*
Timothy	*Phleum pratense*
Common reed	*Phragmites australis*
Ribwort plantain	*Plantago lanceolata*
Greater plantain	*Plantago major*
Annual meadow grass	*Poa annua*
Meadow grass	*Poa pratensis*
Rough meadow grass	*Poa trivialis*
Knotgrass	*Polygonum aviculare*
Silverweed	*Potentilla anserina*
Creeping cinquefoil	*Potentilla reptans*
Barren strawberry	*Potentilla sterilis*
Primrose	*Primula vulgaris*
Selfheal	*Prunella vulgaris*
Blackthorn	*Prunus spinosa*
Bracken	*Pteridium aquilinum*
Meadow buttercup	*Ranunculus acris*
Lesser spearwort	*Ranunculus flammula*
Creeping buttercup	*Ranunculus repens*
Yellow rattle	*Rhinanthus minor*
Dog rose	*Rosa cannina*
Bramble	*Rubus fruticosus*
Raspberry	*Rubus idaeus*
Common sorrel	*Rumex acetosa*
Sheeps sorrel	*Rumex acetosella*
Broad-leaved dock	*Rumex obtusifolius*
Curled dock	*Rumex crispus*
Knotted pearlwort	*Sagina nodosa*
Willow species	*Salix* species
Black bog rush	*Schoenus nigricans*
Deergrass	*Scirpus cespitosus*
Marsh ragwort	*Senecio aquaticus*
Ragwort	*Senecio jacobaea*
Prickly sow thistle	*Sonchus arvensis*
Smooth sow thistle	*Sonchus oleraceus*
Lesser stitchwort	*Stellaria graminea*
Greater stitchwort	*Stellaria holostea*
Chickweed	*Stellaria media*
Devilsbit scabious	*Succisa pratensis*
Snowberry	*Symphoricarpos albus*
Dandelion	*Taraxacum officinale agg*
Marsh dandelion	*Taraxacum palustria*
Wood sage	*Teucruim scorodonia*
Red clover	*Trifolium pratense*
White clover	*Trifolium repens*
Furze	*Ulex europaeus*
Nettle	*Urticia dioica*
Germander speedwell	*Veronica chamaedrys*

Thyme-leaved speedwell	*Veronica serphllifolia*
Field speedwell	*Veronica persica*
Wood speedwell	*Veronica montana*
Bush vetch	*Vicia sepium*
Tufted vetch	*Vicia cracca*
Dog violet	*Viola riviniana*
Water mint	*Mentha aquatica*
Goldenrod	*Solidago virgaurea*
Scarlet pimpernell	*Anagallis arvensis*
Yellow Oat grass	*Trisetum flavescens*
Brooklime	*Veronica beccabunga*
Cucoo flower	*Cardamine pratensis*
Bluebell	*Hyacinthoides non scriptus*
Wild Strawberry	*Fragaria vesca*
Pignut	*Conopodium majus*
Kidney Vetch	*Anthyllis vulnereria*
Ash	*Fraxinus excelsior*
Narrow buckler fern	*Dryopteris carthusiana*
Guelder rose	*Viburnum opulus*
Catsear	*Hypochaeris radicata*
Field pennycress	*Thlaspi arvense*
Tormentil	*Potentilla erecta*
Fumitory	*Fumaria officinalis*
Bog Myrtle	*Myrica gale*
Fragrant orchid	*Gymnadenia conopsea*
Broom	*Cytisus scoparius*
Honeysuckle	*Lonicera periclymenum*
Maidenhair spleenwort	*Asplenium trihomanes*
Greater Birdsfoot	*Lotus pedunculatus*
Hedge woundwort	*Stachys sylvatica*
Juncus species	*Juncus species*
Wall barley	*Hordeum murinum*

Table S2. The bird species recorded during the study and the associated season.

Species	Season
Heron *Ardea cinerea*	Both
Mute swan *Cygnus olor*	Both
Wigeon *Anas penelope*	Winter
Teal *Anas crecca*	Winter
Mallard *Anas platyrhynchos*	Both
Hen Harrier *Circus cyaneus*	Winter
Sparrowhawk *Accipiter nisus*	Both
Buzzard *Buteo buteo*	Winter
Kestrel *Falco tinnunculus**	Winter
Merlin *Falco columbarius*	Winter
Pheasant *Phasianus colchicus*	Both
Moorhen *Gallinula chloropus*	Both
Lapwing *Vanellus vanellus**	Winter
Jack snipe *Lymnocryptes minimus*	Winter
Snipe *Gallinago gallinago*	Both
Woodcock *Scolopax rusticola*	Winter
Curlew *Numenius arquata*	Both
Black-headed gull *Larus ridibundus*	Winter
Stock dove *Columba oenas**	Breeding
Woodpigeon *Columba palumbus**	Both
Collard dove *Streptopelia decaocto*	Winter
Cuckoo *Cuculus canorus*	Breeding
Skylark *Alauda arvensis**	Both
Swallow *Hirundo rustica*	Breeding
Meadow pipit *Anthus pratensis*	Both
Grey wagtail *Motacilla cinerea*	Both
Pied wagtail *Motacilla alba yarrellii*	Both
Wren *Troglodytes troglodytes*	Both
Dunnock *Prunella modularis*	Both
Robin *Erithacus rubecula*	Both
Stonechat *Saxicola torquata*	Both
Blackbird *Turdus merula*	Both
Fieldfare *Turdus pilaris*	Winter
Song thrush *Turdus philomelos*	Both
Redwing *Turdus iliacus*	Winter
Mistle Trush *Turdus viscivorus*	Both
Grasshopper warbler *Locustella naevia*	Breeding
Sedge warbler *Acrocephalus schoenobaenus*	Breeding
Whitethroat *Sylvia communis**	Breeding
Blackcap *Sylvia atricapilla*	Breeding
Chiffchaff *Phylloscopus collybita*	Breeding
Willow warbler *Phylloscopus trochilus*	Breeding
Goldcrest *Regulus regulus*	Both
Long-tailed tit *Aegithalos caudatus*	Both
Coal tit *Parus ater*	Both
Blue tit *Parus caeruleus*	Both
Great tit *Parus major*	Both
Jay *Garrulus glandarius*	Both
Magpie *Pica pica*	Both
Chough *Pyrrhocorax pyrrhocorax*	Winter
Jackdaw *Corvus monedula**	Both
Rook *Corvus frugilegus**	Both
Hooded crow *Corvus corone cornix*	Both
Starling *Sturnus vulgaris**	Both
House sparrow *Passer domesticus*	Both
Chaffinch *Fringilla coelebs*	Both
Brambling *Fringilla montifringilla*	Winter

Greenfinch *Carduelis chloris**	Both
Goldfinch *Carduelis carduelis**	Both
Linnet *Carduelis cannabina**	Both
Redpoll *Carduelis flammea*	Both
Bullfinch *Pyrrhula pyrrhula*	Both
Yellowhammer *Emberiza citrinella**	Both
Reed bunting *Emberiza schoeniclus**	Both

*Farmland Bird Indicator species (Gregory et al. 2004)

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