



PROJECT MUSE®

Does Landscape Structure Affect the Presence of Woodland
Specialist Pollinators in Farmland? Implications for
Agri-Environment Scheme Design

Sara Ruas, Ruth Kelly, Karzan S.D. Ahmed, Stephanie Maher, Ellen O'Hora,
Alessio Volpato, Daire Ó hUallacháin, Michael J. Gormally, Jane C. Stout, James
Moran

Biology and Environment: Proceedings of the Royal Irish Academy, Volume
122B, Number 1, 2022, pp. 17-37 (Article)

Published by Royal Irish Academy



➔ For additional information about this article
<https://muse.jhu.edu/article/854694>

[Access provided at 24 May 2022 16:41 GMT with no institutional affiliation]



This work is licensed under a Creative Commons Attribution 4.0 International License.

Sara Ruas
(Corresponding author;
email: Sara.Ruas@
teagasc.ie. ORCID iD
<https://orcid.org/0000-0002-2940-9892>),
Teagasc, Johnstown
Castle, Co. Wexford,
Ireland, and Galway-
Mayo Institute of
Technology, Marine and
Freshwater Research
Centre, Old Dublin
Rd, Galway, Ireland;
Ruth Kelly, Agri-Food
and Biosciences
Institute, Belfast,
Northern Ireland,
United Kingdom;
Karzan S.D. Ahmed,
National University of
Ireland Galway, Applied
Ecology Unit, Centre for
Environmental Science,
Galway, Ireland;
Stephanie Maher,
Teagasc, Johnstown
Castle, Co. Wexford,
Ireland; Ellen O’Hora,
Trinity College Dublin,
College Green, Dublin 2,
Ireland; Alessio Volpato,
Galway-Mayo Institute
of Technology, Marine
and Freshwater
Research Centre, Old
Dublin Rd, Galway,
Ireland, and Teagasc,
Mellows Campus,
Athenry, Co. Galway,
Ireland; Daire Ó
hUallacháin, Teagasc,
Johnstown Castle,
Co. Wexford, Ireland;
Michael J. Gormally,
National University of
Ireland Galway, Applied
Ecology Unit, Centre for
Environmental Science,
Galway, Ireland; Jane
C. Stout, Trinity College
Dublin, College Green,
Dublin 2, Ireland; James
Moran, Galway-Mayo
Institute of Technology,
Marine and Freshwater
Research Centre, Old
Dublin Rd, Galway,
Ireland.

Received 12 August
2021. Accepted 19
October 2021. Published
16 March 2022.

DOES LANDSCAPE STRUCTURE AFFECT THE PRESENCE OF WOODLAND SPECIALIST POLLINATORS IN FARMLAND? IMPLICATIONS FOR AGRI-ENVIRONMENT SCHEME DESIGN*

Sara Ruas, Ruth Kelly, Karzan S.D. Ahmed, Stephanie Maher, Ellen O’Hora, Alessio Volpato, Daire Ó hUallacháin, Michael J. Gormally, Jane C. Stout and James Moran

ABSTRACT

Some pollinator species found in agricultural areas are strongly dependent on surrounding areas of natural or semi-natural habitats to nest and/or forage. Landscape structure has been shown to influence pollinator communities and understanding how landscape structure affects farmland pollinators can improve Agri-Environment Schemes (AES). This study explored how landscape metrics affect the presence of pollinators associated with woody vegetation in farmland in the Republic of Ireland. Two study regions were selected, and pollinators were collected using pan traps placed in farm linear features. Hoverfly and bee species were selected based on their body size and association with woody vegetation. Relevant landscape structure metrics were extracted from around each trap and used to develop explanatory models for the abundance of pollinators. The total abundance of target species was relatively low but correlated with three explanatory variables: the connectivity of the linear feature to woodlands; the distance from the trap to the closest woodland; and edge density. Hoverfly and bee abundance data, when analysed separately, showed significant differences within regions. Results seem to indicate that incentivising the connectivity of farm linear features to surrounding woodland patches and increasing optimal habitat availability in agricultural landscapes could benefit woodland specialists. This information is helpful to improving AES design.

INTRODUCTION

Pollination is an important ecosystem service underpinning agricultural production and sustainability (MEA 2005; Winfree *et al.* 2008; IPBES 2016). According to Klein *et al.* (2007), 75% of worldwide food crop species benefit from animal pollination and almost 90% of wild flowering plants species benefit, directly or indirectly, from the transfer of pollen by animals (Ollerton *et al.* 2011). In temperate regions, bees (Hymenoptera), hoverflies and other anthophilous flies (Diptera), butterflies (Lepidoptera) and beetles (Coleoptera) are the most important pollinators (Vanbergen and the Insect Pollinators Initiative 2013; Rader *et al.* 2016).

Substantial losses of pollinators have been reported in many regions of the globe and for many European countries it is well documented (e.g. Potts *et al.* 2010). Pollinator declines are especially

high in regions characterised by intensive agriculture (Chagnon 2008; Connelly *et al.* 2015; Kovács-Hostyánszki *et al.* 2017). Agricultural crops and wildlife (e.g. wild plant species) dependent on pollinating insects are therefore at risk (Kearns, Inouye and Waser 1998; Biesmeijer *et al.* 2006; Betts *et al.* 2019). Abundance and diversity of pollinators influence the delivery of pollination services (Winfree *et al.* 2007, 2008). A more species-rich community of pollinators has an increased diversity of functional traits, which can also be beneficial for production (Dainese *et al.* 2019; Pfeiffer *et al.* 2019; Woodcock *et al.* 2019) and provides insurance for future changes in cropping systems (Yachi and Loreau 1999).

Landscape structure and diversity are important factors for maintaining pollinator density and species richness within farming areas (e.g. Bianchi *et al.* 2006; Hendrickx *et al.* 2007; Kennedy *et al.* 2013). Landscape structure is defined as the pattern

**In Memoriam* of Karzan Sabah D. Ahmed, a great researcher and an exceptional human being.

of a landscape, which is determined by the number of different types of use (composition), and also by its configuration, i.e. the size, shape, arrangement and distribution of individual landscape elements (adapted from Walz 2011 and Mitchell *et al.* 2013). Landscape metrics include the density and area of landscape elements/habitats, heterogeneity, evenness, fragmentation and connectivity (Riitters *et al.* 1995; Walz 2011).

Pollinator species found in agricultural areas can be divided into two main groups: a) species that nest and forage in farm habitats (grasslands, orchards, embankments, hedgerows, etc.) and b) species that are dependent on surrounding areas of natural or semi-natural habitats (woodlands, wetlands, etc.) to nest and/or forage.

The spatial organisation of habitats, such as their fragmentation or connectivity in a landscape, also seem to influence survival and dispersal capacity of many pollinating insects (Hadley and Betts, 2012; Viana *et al.* 2012; Boscolo *et al.* 2017; but see Fahrig 2017). For example, patch isolation can influence species richness and density of butterflies (Öckinger and Smith 2006; Herrault *et al.* 2015); distance from natural habitats containing suitable nesting sites can affect pollinator richness and abundance (Kremen *et al.* 2004; Ricketts *et al.* 2008; Jauker *et al.* 2009); connected hedgerows facilitate the movement of bumblebees (Cranmer *et al.* 2011); and the connectivity of these linear elements to forest patches promote the abundance of hoverflies in farmland (Haenke *et al.* 2014). In turn, the degree of isolation of natural areas has a negative effect on this group of pollinators (Herrault *et al.* 2015) and functional connectivity (see for example Taylor *et al.* (2006) for definition) has been reported to affect species richness and abundance of flower-visiting bees (Boscolo *et al.* 2017).

The effects of landscape structure vary depending on species mobility, foraging behaviour and habitat requirements (Steffan-Dewenter *et al.* 2002; Zurbuchen 2010; Popov *et al.* 2017). Thus, the response of organisms to landscape should be investigated at the appropriate scale (Eigenbrod 2016). For example, as pollinators' body size decreases, what would be considered a 'local-scale factor' for highly mobile species might be considered a 'landscape-scale factor' for low-mobility species. Studies report that small-bodied, solitary bee species are more bound to semi-natural habitats because they are less able to disperse, exhibiting shorter foraging distances (Brosi *et al.* 2008; Carrié *et al.* 2017). Therefore, what is considered a fragmented or connected habitat patch will depend on the flight distances that characterise the species (see Wright *et al.* 2015).

To support the abundance and richness of habitat specialists within agroecosystems, a better understanding of the impact of landscape structure is required (Popic *et al.* 2013). This understanding

can be used to improve AES design (Senapathi *et al.* 2017) and support the objectives of the *EU Biodiversity Strategy for 2030* (European Commission 2020a).

This study aims to understand the landscape-related factors (compositional and configurational) that influence the presence of woodland pollinator species in farmland in Ireland. The study focused on species that have lower (predicted) dispersal distances due to small/medium body sizes (Ball and Morris 2015; Falk 2015; Speight *et al.* 2016). Ireland is one of the least wooded countries in Europe (e.g. Bullock and Hawe 2014). However, the remaining small woodland reserves scattered across a landscape have the potential to deliver high levels of multiple ecosystem services (Valdés *et al.* 2019). Since these natural habitats have the potential to support many wild pollinators, they can provide complementary pollination services (Bodin *et al.* 2006; Garibaldi *et al.* 2011, 2013; Krishnan *et al.* 2020). We hypothesise that the abundance of woodland specialists will be higher in landscapes with a higher proportion of woody areas and with higher connectivity between farm and woody habitats.

METHODOLOGY

STUDY AREAS AND FARM SELECTION

Two study areas in Ireland were selected using existing sub-catchments maps (shapefiles created by the Irish Environmental Protection Agency 2018). The study areas were located in contrasting regions (north-west and south-east of Ireland; see Figure 1), with climatic variations. The selected sub-catchment in the north-west (Co. Sligo) has an area of 145.5km², whilst the sub-catchment located in the south-east (Co. Wexford) has an area of 197.12km². Co. Sligo has slightly lower mean temperatures and higher rainfall than Co. Wexford (mean annual temperature and precipitation in Co. Sligo: 9.6 °C and 1260.1mm; and in Wexford 9.8 °C and 840.2mm; Met Éireann 2020). Pollinator data were collected from May to August 2019. Monthly temperature and precipitation for this period are shown in Table A1 in the Appendix.

In each sub-catchment, fifteen farm parcels were selected for sampling pollinators. A farm parcel consisted of two-five adjoining fields and all surrounding linear features. The selection of farm parcels was based on the average habitat quality score from all existing habitats in each farm parcel (i.e. grasslands, hedgerows, drainage ditches, stonewalls or heathlands), which were first scored individually using tailored scorecards developed by Rotchés-Ribalta *et al.* (2021). The scorecards comprised several criteria, such as vegetation structure, cover and abundance of positive plant species and poaching levels (see Rotchés-Ribalta *et al.* 2021 for a detailed description

Cite as follows: Ruas, S., Kelly, R., Ahmed, K.S.D., Maher, S., O'Hara, E., Volpato, A., Ó hUallacháin, D., Gormally, M.J., Stout, J.C. and Moran, J. 2022 Does landscape structure affect the presence of woodland specialist pollinators in farmland? Implications for Agri-Environment Scheme design. *Biology and Environment: Proceedings of the Royal Irish Academy* 2022.

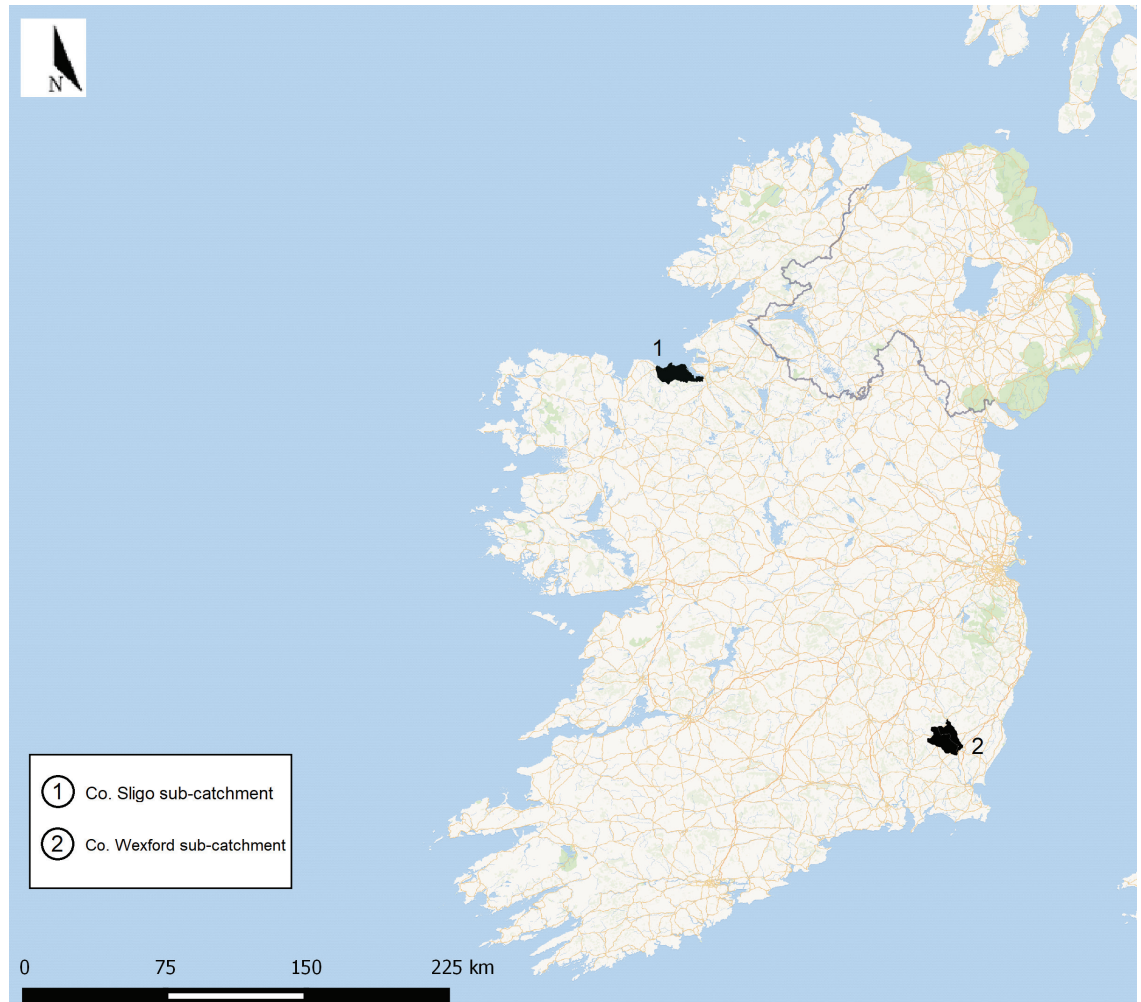


Fig. 1—Location of the study areas (1 – Co. Sligo sub-catchment; 2- Co. Wexford sub-catchment) in the Republic of Ireland.

of habitat quality scoring). The average habitat quality score of each parcel was determined from the scores of each individual farm feature. In each study area, five farm parcels of low-quality score, five farm parcels of intermediate quality score and five farm parcels of high quality score were surveyed.

Pollinator traps had to be at least 1km apart from one another (to reduce the risk of spatial autocorrelation), resulting in one farm parcel in the Co. Sligo study region being excluded. One of the surveyed parcels in Co. Wexford was also excluded since it was located outside of the sub-catchment boundaries and functional connectivity metrics could not be determined. In total, 28 farm parcels were considered. In Co. Sligo, all selected farms were dedicated to livestock production (grasslands and semi-natural pastures, such as heathlands). In Co. Wexford, besides livestock production, five of the selected farm parcels had fields of cereals and other crops (e.g. barley, radish).

POLLINATOR SAMPLING

Insect samples were collected at approximately three-weekly intervals from May to August 2019 (five sampling rounds in total) using sets of coloured pan traps (Stanley *et al.* 2013; Power *et al.* 2016). Each set of pan traps set consisted of three fluorescent, pre-painted plastic bowls (375ml capacity, 135mm diameter), one of each colour (blue, white and yellow) (Moreira *et al.* 2016). In each parcel, three sets of pan traps were installed (nine bowls in total) along a single farm linear feature. Each set of pan traps was placed 15m apart from each other and at least 10m away from the end of the linear feature. The pan traps were partly filled with water with a drop of soap to break the surface tension. Traps were collected after 48 hours and all captured specimens per site were pooled and stored in 70% ethanol for later identification (Stanley *et al.* 2013). Pollinating insects (solitary bees, bumble bees, hoverflies) were subsequently identified to species level, using Ball and Morris (2015) and Stubbs and Falk (2002) for

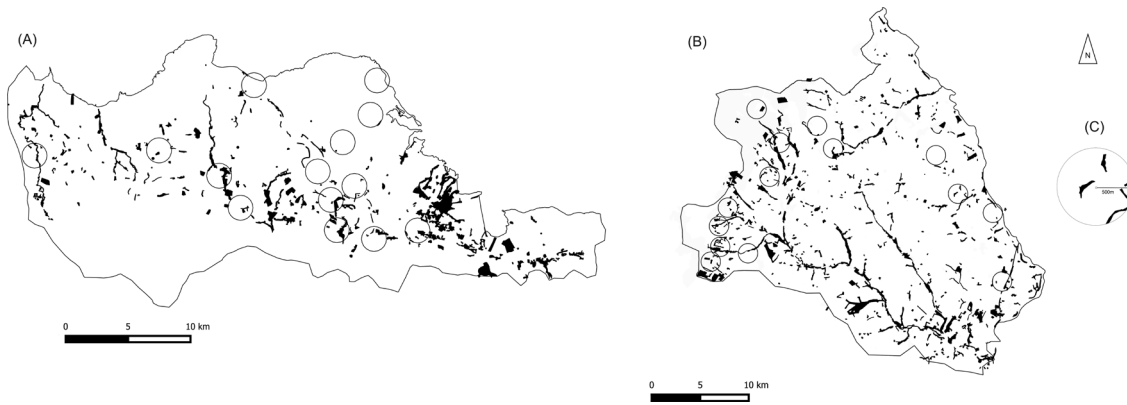


Fig. 2—(A): ‘Optimal habitat’ map for Co. Sligo; (B): ‘Optimal habitat’ map for Co. Wexford. The outlined circles represent the 500m buffers around the centre of the linear feature where the traps were placed. (C): Example of a buffer of 500 m from the pan trap locations, with optimal habitat shown in black.

Syrphidae; Falk (2015) and Else and Edwards (2018) for bees. The abundance of each species per parcel was calculated as the sum of the individuals found in the traps over the five sampling rounds.

From the insects collected, a subset of species was selected, based on two criteria:

1) Body size/mobility: only species with body sizes less than or equal to 10mm (as defined by Moquet *et al.* 2018) were selected. Species with body length above this threshold were considered highly mobile and therefore excluded from the present analysis. Body size is reported to be an important factor for hoverfly and bee dispersal (Greenleaf *et al.* 2007; see for example Aguirre-Gutiérrez *et al.* 2016 and Steffan-Dewenter and Tscharncke 1999), thus mobility was inferred taking this information into consideration. Body size length (average between females and males) was determined for each species based on Speight *et al.* (2016) for hoverflies and on Falk (2015) for bees.

2) Woodland specialists: all the species fitting the previous criteria were further classified as ‘habitat generalists’ or ‘habitat specialists’ by consulting existing literature (Ball and Morris 2015 for hoverflies; Falk 2015 for bees). In the end, eight hoverfly species (*Cheilosia albitarsis* (Meigen), *Dasysyrphus albostrigatus* (Fallén), *Dasysyrphus venustus* (Meigen), *Ferdinandea cuprea* (Scopoli), *Melanostoma scalare* (Fabricius), *Meliscaeva auricollis* (Meigen), *Meliscaeva cinctella* (Zetterstedt) and *Platycheirus scutatus* (Meigen)) plus two wild bee species (*Andrena fucata* Smith and *Andrena lapponica* (Zetterstedt)), reported to have preference for woodland habitats, were considered in this analysis.

LANDSCAPE METRICS

Complete habitat maps were developed (remotely) for each sub-catchment on ArcGIS.10.5 (ESRI,

2016) using Google Earth imagery. All areal and linear habitats were digitised and two distinct vector shapefiles for each sub-catchment were generated. We followed Fossitt (2000) level 2 classification for habitat classification and used Smith *et al.* (2011) for determining areal features and linear features minimum mapping sizes. The maps generated for each sub-catchment are available with corresponding Fossitt codes (linear habitats Co. Wexford: doi:10.13140/RG.2.2.31314.79044; linear habitats Co. Sligo: doi:10.13140/RG.2.2.12020.99201; areal habitats Co. Wexford: doi:10.13140/RG.2.2.32153.65128; areal habitats Co. Sligo: doi:10.13140/RG.2.2.20409.60005) and the list of the all habitats and corresponding Fossitt (2000) code can be seen in Table A3 (Appendix). The percentage of each areal and linear habitats (at level 2 of Fossitt 2000) was calculated in each sub-catchment, and the density of linear features was also determined for both regions by dividing the total length of linear feature by the total area of the sub-catchment (results in Figure A1 (Appendix)).

A 500m buffer (in ArcGIS.10.5) was created around each trap (from the middle of the linear feature where the traps were placed) to extract relevant landscape composition and landscape configuration metrics. This choice of spatial extent allowed us to avoid spatial auto-correlation and is in line with other studies (e.g. Joshi *et al.* 2016).

Some of the landscape metrics were extracted from the maps with all habitat types (see Table A3), whilst others were obtained from ‘optimal habitat’ maps – these maps were generated for each sub-catchment by only retaining areas of woody vegetation using the select by attributes function in ArcGIS. Therefore, two separate maps were generated where: background = non-woody vegetation areas and foreground = woody vegetation areas. See Figure 2 (A) and (B) with the

Table 1—List of habitats considered for generating the ‘optimal habitat’ maps (Figure 2), with respective code as in Fossitt (2000).

<i>Habitat</i>	<i>Fossitt (2000) level 2 code</i>
Native woodlands	WN
Native woodlands mixed with planted forest	WN/WD
Scrubland	WS
Hedgerows and treelines with a width > 4m	WS
Mixed areas of native woodland and scrubland	WN/WS
Mixed areas of native woodland or scrubland and wetlands	WN/GM or WS/GM WN/FW or WS/FW
Mixed areas of native woodland or scrubland and semi-natural grasslands	WN/GS or WS/GS
Mixed areas of native woodland or scrubland and heathlands or peatlands	WN/HH or WS/HH WN/PB or WS/PB
Mixed areas of native woodland or scrubland and bracken	WN/HD or WS/HD

generated maps for both Co. Sligo and Co. Wexford and Figure 2 (C) with an illustration of a 500m buffer of ‘optimal habitat’. The habitats assumed to be optimal for the selected hoverfly and bee species (and thus used to generate the ‘optimal habitat’ maps) included native woodland, mixed habitat containing native woodland, and scrub (Table 1). Hedgerows and treelines were not considered in the ‘optimal habitat’ maps, but were considered as a potential explanatory variable (i.e. total length of hedges in a 500m buffer) and thus tested as potential suitable habitat. Conifer plantations were also excluded as an optimal habitat in this study since in Ireland the management is very intensive. A study conducted by Gittings *et al.* (2006) showed that hoverflies richness in conifer plantations is more related to the existence of open areas, broad-leaved trees, shrubs and wet habitat features that surround the plantation than to the habitat created by the conifers *per se*. Consequently, conifer plantations were considered an intensively managed habitat.

All the landscape metrics were divided into *landscape composition metrics* and *landscape configuration metrics*. The compositional metrics calculated for each selected farm parcel were the following: 1) total area of optimal habitat in the 500m buffer; 2) area of intensive production in the 500m buffer; 3) area of semi-natural pastures (heathlands, peatlands and/or mosaics of heathland/semi-natural grasslands etc.) in the 500m buffer; 4) area of semi-natural grasslands in the 500m buffer; 5) total length of hedgerows in the 500m buffer; 6) area of the closest patch of optimal habitat (from the traps); and 7) greatest patch area in the 500m buffer. The configurational metrics generated were: 1) distance from the traps to optimal habitat; 2) connectivity of linear feature where the traps were placed to optimal habitat (i.e. if it was physically linked to an optimal habitat area); 3) production fields average size; 4) edge length in the 500m buffer; 5)

edge density in the 500m buffer; 6) Splitting Index S (SPLI) for the 500m buffer; 7) Landscape Division Index (LDI) for the 500m buffer; 8) Patch Cohesion Index for the 500m buffer; and 9) functional connectivity metrics – comprised of seven indices (dPA, dF, dFWA, dPC, dpCcontra, dpCflux, dPCcon) and determined considering the total extent of the ‘optimal habitat’ maps for each sub-catchment. All variables are explained in detail in Table A2 (Appendix).

Beside these landscape structure variables we considered three additional variables: two variables related to the habitat quality (determined by Rotchés-Ribalta *et al.* 2021) of the parcel where the traps were placed – *habitat quality of fields* and *habitat quality of linear features*; and a categorical variable, *Region*, so we could account potential larger scale landscape differences between the two study regions.

STATISTICAL ANALYSES

Regression models were conducted to investigate the response of pollinator abundance data to the landscape metrics selected. Collinearity amongst the landscape metrics was assessed using a Spearman correlation analysis. When the correlation between two explanatory variables was greater than $|r| \geq 0.7$ (Dormann *et al.* 2013), we decided to run the models, initially, with the variable that provided a more direct explanation (e.g. *Area of the closest woodland patch* instead of the connectivity index *dPC*). The list of variables introduced in the first iteration of the models, corresponding correlated variables and descriptive statistics are shown in Table 2.

Spatial autocorrelation of woodland specialist abundance was investigated via Moran’s I for both sub-catchments combined and separately. Since the results of this analysis showed that the spatial distribution of the pollinator abundance is likely to be a

Table 2—Descriptive statistics of the explanatory variables introduced in the models and correspondent correlated variables ($|r| \geq 0.70$).

<i>Explanatory variables introduced in the model</i>	<i>Descriptive statistics</i>	<i>Correlated variables</i>
<i>Area of habitats of intensive production (in 500m buffer) (m²)</i>	Min: 105,569 Max: 783,953 Median: 613,067 Mean: 561,231	Area of semi-natural pastures (in 500m buffer); total length of hedgerows (in 500m buffer).
<i>Area of semi-natural grasslands (in 500m buffer) (m²)</i>	Min: 0 Max: 248,690 Median: 27,395 Mean: 52,715	None
<i>Closest optimal habitat area (m²)</i>	Min: 1,448 Max: 180,147 Median: 9,743 Mean: 21,841	dA50; dAWF50; dPC50; dPCcintra50; dA250; dAWF250; dPC250; dPCcintra250; dPCFlux250; dA500; dAWF500; dPC500; dPCcintra500; dPCcon500
<i>dF500*</i>	Min: 0.000 Max: 2.019 Median: 0.402 Mean: 0.591	dF50; dF250
<i>Distance to the closest optimal habitat (m)</i>	Min: 0.00 Max: 1631.00 Median: 148.840 Mean: 244.34	None
<i>dPCcon500**</i>	Min: 0.000 Max: 3.973 Median: 0.027 Mean: 0.274	dPCcon50; dPCcon250
<i>Optimal habitat area (in 500m buffer) (m²)</i>	Min: 0 Max: 125,826 Median: 25,122 Mean: 42,957	Edge length (in 500m buffer); edge density (in 500m buffer); greatest patch area (in 500m buffer); landscape Division Index (LDI)
<i>Connectivity of linear feature to optimal habitat</i>	Yes: 15 No: 13	None
<i>Production fields average size (in 500m buffer) (m²)</i>	Min: 8003 Max: 23,210 Median: 12,509 Mean: 13,494	None
<i>Region</i>	Wexford: 14 Sligo: 14	Habitat quality fields
<i>Habitat quality linear features (0-1)</i>	Min: 0.287 Max: 0.565 Median: 0.421 Mean: 0.419	None

*dF: One of the three dIIC fractions estimating the amount of dispersal fluxes between a particular patch (as the origin or destination of those fluxes) and the rest of the patches in the landscape (Saura and Rubio 2010);
**dPCcon: One of the three fractions of dIIC measuring the contribution of the analysed patch to the connectivity between other patches, as a connecting element or stepping stone between them (Saura and Rubio 2010).

result of random spatial processes (Moran's I for both sub-catchments: Moran's I = -0.086, p > 0.05; for Co. Sligo farms' data: Moran's I = -0.072, p > 0.05; Co.

Wexford farms' data: Moran's I = -0.016, p > 0.05) it was not necessary to apply any corrections to the subsequent models.

To determine the relationship between our explanatory variables and the abundance of woodland pollinators, we first attempted to use Generalized Linear Models (GLMs) with a Poisson response distribution and a log-link function, appropriate for use with count data. A modified version of the `stepAIC()` function from the R package MASS (Venables and Ripley 2002) was used for model selection. This modified function uses the AICc (Akaike's corrected information criterion, applicable for small sample sizes) rather than AIC to select the predictor to add at each step (Read *et al.* 2018). The model selection was set for both 'backward' and 'forward' selection. An additional backward elimination approach to find the most parsimonious model was also applied: the least significant effect is removed when it does not meet the significance level of p-value < 0.05 (Hong and Mitchell 2007). For the evaluation of the models' regression fit we considered the models' AIC. We compared different model iterations by replacing the significant variables with those with which they were correlated, to understand if the model would improve its explanatory power. When we obtained an apparently final model, where all the explanatory variables/parameters were significant (p-value < 0.05), we analysed the model's robustness by: a) testing the model for significance by performing a comparison (ANOVA) between the obtained model and a model where the response variables were fitted against 1; b) testing for *overdispersion* (by dividing the model deviance by the residual's degrees of freedom) (Thomas and the Guidebook Team 2017); and c) checking the adjusted-R² value. When *overdispersion* values above 2 were observed, we developed new models and repeated the process, but this time using a Negative Binomial response distribution (log-link function).

Given the different results obtained (higher abundance of hoverflies in Co. Sligo and of bees in Co. Wexford – see Table 3 in Results section), we developed separate models for each taxonomic group and followed the same model development process and the above-mentioned selection criteria for each.

Models' results validation

Because of the small number of samples considered in this study (28 sampling points) and low abundance of selected pollinators, we tested if the significant relationships obtained between explanatory variables and woodland specialist abundance were a 'chance effect', due to possible overfitting.

Thus, results of the models were validated by fitting the emergent significant explanatory variables (via GLMs) to:

a) The abundance of randomly selected species from the full dataset of pollinating insects collated. In particular, we generated a new response variable by randomly selecting a group of ten species from the total list of pollinators – eight hoverfly and two bee

species (from all species sampled). We were careful to exclude the woodland specialist species selected before conducting the random selection of species;

b) Randomly generated variables with the same distribution (Negative binomial, Poisson) and mean ('null models') as the woodland specialist response variables. The random response variables were obtained using the function `rpois(N = sample size; m = mean)` or `rnbinom(N = sample size; m = mean)`. These R functions generate random numerical lists with a Poisson and negative binomial distribution, respectively. Fifteen 'response variables' were created: five different 'response variables' simulating all abundance data combined (hoverflies and bees); five 'response variables' simulating hoverfly abundance data; and five 'response variables' simulating bee abundance data (following Farine (2017)).

This was a strategy utilised to see whether the relationships obtained for the woodland specialists were spurious or not, by testing if the same variables would emerge as significant for other species (non-woodland specialists) and for randomly generated 'response variables'. Consequently, the results of these null models are not discussed in detail. All statistical analyses were conducted in R version 3.5.2 (R Development Core Team 2013).

RESULTS

ABUNDANCE OF WOODLAND SPECIALISTS (HOVERFLIES AND BEES)

A total of 86 woodland specialists from the 10 target species were captured (see Table 3), with 56 hoverflies and 30 bees. This value represented c 2% of the total abundance of pollinators captured (more than 3,500 bees and hoverflies captured in both sub-catchments). The majority (36) of woodland hoverflies captured were *M. scalare*, mostly in Sligo (24 individuals), whilst the most abundant bee species was *A. fucata*, particularly in Co. Wexford (16 individuals).

The best model obtained for the abundance of woodland specialists included the *Distance to the closest optimal habitat*, the *Connectivity of linear feature to optimal habitat* and *Edge density* (of woody areas in a 500m buffer) as significant variables (see Table 4 (A) for model results). The *Connectivity of linear feature to optimal habitat* had a positive relationship with the abundance of woodland specialists, whilst *Edge density* and *Distance to the closest optimal habitat* had a negative relationship with the abundance of these pollinators.

When comparing this model with the model fitted to the abundance of ten randomly selected species (i.e. the null models), none of these three variables had a significant correlation with their abundance. Results of the null models (random

Table 3—List of selected pollinators (with correspondent abundance and number of occurrences) with body size $\leq 10\text{mm}$ (average between males and females) and reported to have preference for woodland habitats.

<i>Species</i>	<i>Habitat/ecology</i>	<i>Abundance</i>	<i>Occurrence</i>	<i>Average body size (mm)</i>
Hoverflies				
<i>Cheilosia albitarsis</i>	Edges of forest clearings and tracks; along hedges.	Total = 10 Co. Sligo = 10 Co. Wexford = 0	Total = 5 Co. Sligo = 5 Co. Wexford = 0	9
<i>Dasysyrphus albobristatus</i>	Forest; most types of coniferous and deciduous forest and conifer plantation, up to the lower limits of the alpine zone.	Total = 4 Co. Sligo = 2 Co. Wexford = 2	Total = 4 Co. Sligo = 2 Co. Wexford = 2	10
<i>Dasysyrphus venustus</i>	Woodland species that occurs in both deciduous and coniferous woodlands; also, along clearings and track sides. Arboreal.	Total = 1 Co. Sligo = 0 Co. Wexford = 1	Total = 1 Co. Sligo = 0 Co. Wexford = 1	8.5
<i>Ferdinandea cuprea</i>	A woodland species (mainly deciduous), frequent found at the edge of clearings and along tracks.	Total = 2 Co. Sligo = 2 Co. Wexford = 0	Total = 2 Co. Sligo = 2 Co. Wexford = 0	9.5
<i>Melanostoma scalare</i>	Most types of humid/mesophilous forest (both coniferous and deciduous), but also in more open situations; along hedges in various sorts of farmland, parks and along track sides in conifer plantations.	Total = 24 Co. Sligo = 16 Co. Wexford = 8	Total = 12 Co. Sligo = 8 Co. Wexford = 4	8
<i>Meliscaeva auricollis</i>	Many types of forest (both deciduous, broad-leaved evergreen and coniferous) and conifer plantations.	Total = 9 Co. Sligo = 4 Co. Wexford = 5	Total = 6 Co. Sligo = 3 Co. Wexford = 3	9.5
<i>Meliscaeva cinctella</i>	A widespread woodland species. Found in deciduous and coniferous forest; also, in hedgerows and suburban gardens and parks.	Total = 4 Co. Sligo = 4 Co. Wexford = 0	Total = 2 Co. Sligo = 2 Co. Wexford = 0	10
<i>Platycheirus scutatus</i>	Forest; most types of deciduous forest, especially scrub woodland; also along field hedges, in fruit and olive orchards, in suburban gardens and parks and in conifer plantations.	Total = 2 Co. Sligo = 2 Co. Wexford = 0	Total = 2 Co. Sligo = 2 Co. Wexford = 0	8
Bees				
<i>Andrena fucata</i>	Woodland and scrub; hedgerows; heathland.	Total = 20 Co. Sligo = 4 Co. Wexford = 16	Total = 11 Co. Sligo = 2 Co. Wexford = 9	8
<i>Andrena lapponica</i>	Woodland and scrub, heathland.	Total = 10 Co. Sligo = 2 Co. Wexford = 8	Total = 7 Co. Sligo = 2 Co. Wexford = 5	8.5

negative binomial 1, 2, 3, 4 and 5) reveal that for only one of the models (random negative binomial 2), *Edge density* emerges as a significant explanatory variable, but this time with a positive correlation

with the randomly generated variable. However, this model has a higher AIC value and a lower-adjusted R^2 . The fact that the variables never emerge as significant for any other of the tested null models, and

Table 4—(A): Results of the model obtained for the abundance of woodland specialists (hoverflies and bees) with respective results for goodness of fit for the abundance of 10 randomly selected species and for the random negative binomial 1,2,3,4 and 5; (B): Results of the model obtained for the abundance of hoverflies with respective results for goodness of fit for the abundance of 8 randomly selected hoverflyspecies and for the random negative binomial 1,2,3,4 and 5; (C): Results of the model obtained for the abundance of bees with respective results for goodness of fit for the abundance of 2 randomly selected beespecies and for the random POISSON 1, 2,3,4 and 5. Significance levels: ‘**’ 0.001 ‘***’ 0.01 ‘**’ 0.05.**

(A) WOODLAND SPECIALISTS (HOVERFLIES AND BEES) MODEL RESULTS (Model Type: Negative Binomial)			
<i>Variable</i>	<i>Overdispersion test results</i>	<i>Explanatory variables (significant)</i>	<i>Model goodness of fit</i>
Abundance of woodlands specialists N=28; Mean = 3.321	1.235	Distance to closest optimal habitat=-0.002* Edge density = -55.534* Connectivity of linear feature to optimal habitat (Yes) =0.836*	AIC = 130.02 Adjusted R ² = 0.318 χ^2 (anova) <0.050***
Model Validation Results (Models Type: Negative Binomial)			
Abundance of 10 randomly selected species N=28; Mean = 38.79	1.211	None	AIC = 263.87 Adjusted R ² = 0.000 χ^2 (anova) > 0.050
Random Negative Binomial 1 N=28; Mean = 3.321	1.267	None	AIC = 130.81 Adjusted R ² = 0.051 χ^2 (anova) > 0.050
Random Negative Binomial 2 N=28; Mean = 3.321	1.300	Edge density = 94.235**	AIC = 134.44 Adjusted R ² = 0.261 χ^2 (anova) <0.050*
Random Negative Binomial 3 N=28; Mean = 3.321	1.242	None	AIC = 140.12 Adjusted R ² = 0.173 χ^2 (anova) > 0.050
Random Negative Binomial 4 N=28; Mean = 3.321	1.253	None	AIC = 101.28 Adjusted R ² = 0.188 χ^2 (anova) > 0.050
Random Negative Binomial 5 N=28; Mean = 3.321	1.316	None	AIC = 118.99 Adjusted R ² =0.159 χ^2 (anova) > 0.050
(B) HOVERFLIES (Model Type: Negative Binomial)			
Abundance of woodlands specialists N=28; Mean = 2.857	1.241	Region (Wexford) = -0.957* Connectivity of linear feature to optimal habitat (Yes) =1.091*	AIC = 106.51 Adjusted R ² =0.365 χ^2 (anova) <0.050*
Model Validation Results (Models Type: Negative Binomial)			
Abundance of 8 randomly selected species N=28; Mean = 20.00	1.275	Region (Wexford) = 2.203***	AIC = 197.08 Adjusted R ² =0.377 χ^2 (anova) <0.050*
Random Negative Binomial 1 N=28; Mean = 2.857	1.173	None	AIC = 108.83 Adjusted R ² =0.087 χ^2 (anova) >0.050

Random Negative Binomial 2 N=28; Mean = 2.857	1.308	None	AIC = 116.71 Adjusted R ² =0.179 χ^2 (anova) >0.050
Random Negative Binomial 3 N=28; Mean = 2.857	1.233	None	AIC = 137.48 Adjusted R ² = 0.064 χ^2 (anova) >0.050
Random Negative Binomial 4 N=28; Mean = 2.857	1.182	None	AIC = 118.54 Adjusted R ² = 0.101 χ^2 (anova) > 0.050
Random Negative Binomial 5 N=28; Mean = 2.857	1.246	None	AIC = 134.59 Adjusted R ² = 0.064 χ^2 (anova) > 0.050
(C) BEES (Model Type: Poisson)			
Abundance of woodlands specialists N=28; Mean = 2.857	1.693	Region (Wexford) = 1.642	AIC = 86.229 Adjusted R ² = 0.258 χ^2 (anova) < 0.050***
Model Validation Results (Models Type: Poisson)			
Abundance of 2 randomly selected species N=28; Mean = 2.857	1.254	None	AIC = 219.770 Adjusted R ² = 0.102 χ^2 (anova) < 0.050***
Random Poisson 1 N=28; Mean = 2.857	0.728	None	AIC = 73.22 Adjusted R ² = 0.000 χ^2 (anova) > 0.050
Random Poisson 2 N=28; Mean = 2.857	1.138	None	AIC = 78.761 Adjusted R ² = 0.000 χ^2 (anova) > 0.050
Random Poisson 3 N=28; Mean = 2.857	1.636	None	AIC = 88.449 Adjusted R ² = 0.000 χ^2 (anova) > 0.050
Random Poisson 4 N=28; Mean = 2.857	0.853	None	AIC = 71.223 Adjusted R ² = 0.096 χ^2 (anova) > 0.050
Random Poisson 5 N=28; Mean = 2.857	1.457	None	AIC = 89.790 Adjusted R ² = 0.000 χ^2 (anova) > 0.050

for the abundance of ten randomly selected species, indicates that the model results for the abundance of woodland specialist are likely to be more than a 'chance effect'.

When we analysed the relation between the explanatory variables (Table 2) and the abundance of woodland hoverflies (GLM with negative binomial distribution), we verified that *Region* and the *Connectivity of linear feature to optimal habitat* emerged as significant, with less abundance of these pollinators found in Co. Wexford and a higher abundance found in the farm-linear features connected to optimal habitat (i.e woody areas). When comparing this model with the model fitted to the abundance of

eight randomly selected hoverfly species, we can see that this time there was a higher abundance of these randomly selected species in Co. Wexford and that there was no relation between these species abundance and the *Connectivity of linear feature to optimal habitat*. Additionally, results of the null models (random negative binomial 1, 2, 3, 4 and 5) reveal that neither of the two significant variables ever emerged as significant for the randomly generated response variables. Thus, the model results for the abundance of hoverfly woodland specialists are likely to be more than a 'chance effect' (all results shown in Table 4 (B)).

For bees, from all variables inputted in the model, only *Region* emerged as significant in the

GLM (with a Poisson distribution). In contrast to what was observed for hoverflies, a higher abundance of the two woodland bee species selected was sampled in Co. Wexford. This variable did not emerge as related to the abundance of the two randomly selected bee species, or to any of the randomly generated Poisson variables (Poisson 1, 2, 3, 4, and 5), which indicates that this variable is likely to have a 'real' significant effect on the abundance of woodland bee species (results shown in Table 4 (C)).

Since *Region* was a significant variable in both hoverfly and bee models, we also examined whether there were significant differences between sub-catchments, via Kruskal-Wallis tests, for all the explanatory variables considered in the model initially (e.g. between the total area of optimal habitats within the 500m buffers defined around each trap; between the distance to the closest optimal habitats, etc.). We did not find significant differences between regions for any of the variables (Kruskal-Wallis $p > 0.050$), except for the area of semi-natural pastures, which was higher in the 500m buffers surrounding the Co. Sligo farm parcel. Results from this analysis are presented in the Table A4 (Appendix). Whilst they are not discussed in detail, they enable better interpretation of the model results for the abundance of the two groups of woodland pollinators.

DISCUSSION

This study aimed to explore how landscape structure is related to the presence of low-mobility woodland pollinators in Irish farmland. Few explanatory variables explained the patterns observed; however, it was possible to verify that some of the selected landscape metrics had a significant relationship with the abundance of woodland pollinators.

The abundance of woodland pollinators collected was relatively low, with two species (one hoverfly: *M. scolare*; and one bee, *A. fucata*) dominating. This might be related to the fact that these two species are less exclusively bounded to woodland areas and are considered as frequently occurring (Ball and Morris 2015; Falk 2015). Yet, the abundance of these two species is also not very high, considering the number of traps placed in the farms, the amount and variability of farms sampled and the sampling period distribution. The low abundance can be seen as a result in itself and is not necessarily surprising, due to an overall low cover of native woodland in the Irish landscape (Bullock and Hawe 2014). In this study, we also verified a low percentage of area of woody habitats in the 500m buffers of our sampled farms (5.5% of the total buffer area on average) with most of the land-use dedicated to food and fibre production. In fact, most of the area in both study regions, at sub-catchment level, comprises improved grasslands

(almost 44% in Co. Wexford and 54% in Co. Sligo) (see Figure A1 in the Appendix).

It could be possible that the low abundance of woody vegetation areas is compensated by the presence of hedgerows. Yet, the total length of hedgerows in the surrounding 500m from the traps did not correlate with the woodland specialist abundance. This might indicate that these linear farmland habitats can function as corridors and provide resources (Cranmer, McCollin and Ollerton 2011), but their ecological role does not seem to be equivalent to that of areas of woody vegetation. This could be the result of management actions that reduce hedgerow ecological value (Byrne and del Barco-Trillo 2019).

LANDSCAPE CONFIGURATIONAL METRICS HAVE A GREATER INFLUENCE THAN SELECTED COMPOSITIONAL METRICS

We found a positive relationship between the abundance of woodland specialists, particularly hoverflies, and connectivity of the linear feature (where the traps were placed) to a woodland patch, which is in line with other studies (e.g. Ricketts *et al.* (2006) for different pollinator communities; Van Geert *et al.* (2010) for pollen dispersal; Haenke *et al.* (2014) and Jovičić *et al.* (2017) for hoverflies; and Schirmel *et al.* (2016) for predatory carabids). Thus, besides retaining and/or creating linear features to increase farmland biodiversity (e.g. Sullivan *et al.* 2017), it may also be important to develop a network of connected linear elements to areas of existing or new semi-natural habitats. This can potentially result in an increase in pollination services in farmland (Cranmer, McCollin and Ollerton 2011; Delattre, Vernon and Burel 2013). However, connectivity of woodland patches to woody areas was not related to the abundance of selected bee species. In other studies, e.g. Krewenka *et al.* (2011), grass strips acted as sinks rather than corridors for bees, and connectivity was not considered important. This result might support the idea that species and/or taxonomic groups respond differently to landscape structure (e.g. Kennedy *et al.* 2013; Moquet *et al.* 2018). In fact, specific functional traits among pollinator groups might explain distributional variability (Aguirre-Gutiérrez *et al.* 2016). Yet, because of the low abundance values when we separated both groups, the observed patterns for the separate regression models can be considered less reliable.

An increase in distance from natural habitats has been reported to affect pollinator richness and abundance in agricultural areas in other studies, since fewer pollinator species are able to either forage or nest in agricultural areas (Jauker *et al.* 2009; Garibaldi *et al.* 2011; Bailey *et al.* 2014). Similarly, we observed that the abundance of woodland pollinators in our farms decreased with increasing distance to optimal habitat patches. Moquet *et al.* (2018) also showed that distance to forests areas was

the main factor related to hoverfly composition in their study, but this variable did not emerge as significant when we considered hoverfly and bee abundance data separately, again, possibly due to the low abundance values.

The functional connectivity metrics determined did not emerge as significant. However, these functional connectivity metrics are only *potential* (based on theoretical flight distances and theoretical ideal habitats) and not *actual* (Calabrese and Fagan 2004). In fact, foraging distances might vary with the environmental conditions of the other landscape units (e.g. physical resistance of the different habitats to flight, in Ricketts, 2001). Other studies have reported that the quality of different land-use units may change pollinator flux among patches, and that they do not perceive landscapes as binary habitat-non-habitat systems (Abdel Moniem *et al.* 2013; Slancarová *et al.* 2014). This influences how functional connectivity metrics are related to pollinator movement (Slancarová *et al.* 2014). Thus, we cannot conclude that there is no effect of these variables *per se*: we can only state that we did not find any relationship based on our assumptions, in particular the flight distances.

A negative relationship between the combined abundance of woodland specialists and edge density (compositional metric) was observed (but not for hoverfly and bee abundance separately). This variable was correlated with total area of optimal habitats and edge length in the 500m buffers in our study, yet, when we replaced edge density by either the total area of optimal habitats or edge length, the model lost explanatory power, which suggests variable effects of these three factors on the abundance of woodland specialists. Positive relationships between edge habitat in woodland patches and the abundance of pollinators have been reported by Popov *et al.* (2017) and Roberts *et al.* (2017). Nonetheless, we believe that our results are not contradictory to this observation: we sampled fewer woodland pollinators in farms when there was a high edge density of optimal habitat in the surrounding 500m, possibly because these species dwell more in the edge than in the surrounding agricultural fields (where we sampled) due to higher ratios of feeding resources and nesting sites available. In fact, a great number of the species that were selected in this study are associated with edge habitats of woodlands and forest areas (see Table 3). Thus, this result might indicate that the abundance of woodland specialists is a function of high levels of edge habitats per total forest area.

Variables related to habitat quality for pollinators in the surrounding woodland patches and farmland habitats (flower resources, nesting sites, and vegetation structure) were not considered in this study, since our goal was to contribute to understanding the effects of landscape structure in the abundance of woodland specialists in farmland.

However, this information could improve models' performance and could allow the disentangling of some of the results obtained, since habitat quality (for pollinators) of surrounding habitats can affect pollinator abundance (Krewenka *et al.* 2011; Kennedy *et al.* 2013).

REGIONAL DIFFERENCES BETWEEN HOVERFLIES AND BEES

When hoverfly and bee abundance data were separated, most landscape metrics lost significance and the variable *Region* consistently emerged as significant. Whilst the abundance of woodland hoverflies was significantly lower in Co. Wexford, the abundance of woodland bees was lower in Co. Sligo. The existence of contradictory responses between these pollinator groups, and pollinator guilds in general, is reported in other studies (e.g. Jauker *et al.* 2009; Robert *et al.* 2017), thus it is not surprising. In the case of the bees, *Region* was the only significant predictor, but its real effect should be carefully considered since this model had the lowest adjusted-R² value.

We could expect some underlying regional differences between the landscape variables introduced in the models (Table 2). However, no significant differences between most of the configurational and compositional metrics were found between the two regions, except for the fact that the amount of semi-natural pastures in the 500m buffers was higher in Co. Sligo. In fact, the percentage of semi-natural pastures represents around 20% of the total area in Co. Sligo (Figure A1, Appendix) and a much lower percentage in Co. Wexford (circa 5.7%). Yet, if the area of semi-natural pastures explained the contradictory pattern observed, then it would most likely have emerged as a significant predictor, which was not the case. Thus, it seems that other regional differences are affecting the abundance of hoverflies and bees and were not captured in this study.

Hence, we put forward two possible hypotheses: 1) the regional effect observed can be related to differences in quality of woodland patches for both pollinator groups (and therefore pollinator abundance in the woodland areas), such as the degree of wetness or flower resources available (see, for example, Fuller *et al.* 2017). Willaert (2019) found that the attractiveness of plant species differed largely between wild pollinator groups and that flower presence at the local scale affected pollinator species richness and abundance more than correlated landscape structure variables. However, we do not have information on pollinator abundance in the woodland areas or woodland patch quality from our study to test this hypothesis. Alternatively, 2) the low abundance data in the two taxonomic groups reduced the statistical performance of the models. In fact, when we randomly selected eight hoverfly

species from the total sampled species list, we verified that Co. Wexford had a positive relationship with their abundance, indicating that not all hoverflies are negatively affected by existing regional differences. Furthermore, when we randomly selected two other bee species, we found that there was no difference between regions.

This difference in pattern of occurrence between bees and hoverflies could be related to differences in weather conditions between study regions (see Table A1 in the Appendix). However, we would then expect a higher abundance of both groups in Co. Wexford due to higher average temperatures and lower precipitation levels (Totland 1994; Lawson and Rands 2019); yet, the species of hoverfly considered were more abundant in Co. Sligo, which would imply that these species prefer lower temperatures and higher precipitation levels.

Although the apparently contradictory effect of region on the species selected might indicate that some parameter choices valid in one environment may not hold when other landscapes and environments are considered (Newman *et al.* 2019), this is a tentative suggestion given the low abundance of insects in our analysis. Therefore, as a tentative conclusion, we consider that *Region* emerging as explanatory variable might be a combination of a low sample size when the groups are separated, with some larger sub-catchment ecological differences that we could not disentangle (such as the quality of woodland areas and/or intensity of agricultural activities), which would have to be affecting the selected species of both groups and not the taxonomic groups *per se*.

IMPLICATIONS FOR AES DESIGN

Even if most of the agricultural production in both study regions is not significantly dependent on pollination (because it comprises mainly grasslands and cereal production), future climate change might lead to alterations in farming systems and the need for adapting to new weather conditions. In fact, climate change is projected to improve the conditions for growing crops in northern Europe (European Environment Agency 2019). Thus, the occurrence of a diverse pollinator community in farmlands, including woodland pollinators, can provide insurance for future production (Yachi and Loreau 1999; Winfree *et al.* 2007, 2008) and increase productivity of crops (Dainese *et al.* 2019; Pfeiffer *et al.* 2019; Woodcock *et al.* 2019). Furthermore, multispecies swards (a key strategy for reducing chemical fertiliser input and related greenhouse gas emissions) (Peyraud *et al.* 2010) comprise some species (e.g. clover) with seed set rates depending on pollination (e.g. Jing *et al.* 2021).

A diverse community of pollinators can also have positive effects on the ecological health of

ecosystems, since pollinator richness can be important for the population dynamics of wild plant species (e.g. Ollerton *et al.* 2011; Albrecht *et al.* 2012). Hence, the presence of the woody areas scattered across the agricultural landscape (particularly when connected through farmland linear features) has the potential to maintain some wild pollinator populations, acting as a reservoir and source (Taki, Kevan, and Ascher 2007; Gaytán *et al.* 2020). These pollinators then travel to farmland to acquire floral resources and, thus, support pollination services (e.g. Carvalheiro *et al.* 2011; Valdés *et al.* 2019; Huais *et al.* 2020).

Despite the limitations and complexities of our results, they indicate that the safeguarding of pollination services by woodland specialists in farmlands might depend on the connectivity of linear features to surrounding woodland patches, and on maintaining and creating small woodland patches around or within the farms, thus reducing the distance from the fields to woodland areas. Semi-natural woodlands and scrublands occurring in farmlands are considered undervalued habitats and ineligible for AES in Ireland (Rotchés-Ribalta *et al.* 2021). Therefore, AES that aim to increase the abundance and diversity of pollinators in farmland (as highlighted in the recent *Farm to Fork Strategy* (European Commission 2020b)), should be incentivising: a) a decrease in distance from production fields to semi-natural areas, by increasing (and incentivising farmers to retain) the quantity of woodlands and scrublands within farmlands; and b) the creation and connection of farmland linear features to these areas.

ACKNOWLEDGEMENTS

The authors thank all farmers and advisors who collaborated with the project; and Julien Carlier for his help and advice in the development of the functional connectivity metrics. This work was supported by the Irish Department of Agriculture, Food and the Marine (DAFM) (RSF15_S_619).

REFERENCES

- Abdel Moniem, H., Eldien, M. and Holland, J.D. 2013 Habitat connectivity for pollinator beetles using surface metrics. *Landscape Ecology* **28**, 1251–67. doi:10.1007/s10980-013-9886-9.
- Aguirre-Gutiérrez *et al.* 2016 Functional traits help to explain half-century long shifts in pollinator distributions. *Scientific Reports* **6**, 24451. doi:10.1038/srep24451.
- Albrecht *et al.* 2012 Diverse pollinator communities enhance plant reproductive success. *Proceedings of the*

- Royal Society B: Biological Sciences* **279** (1748), 4845–52. doi:10.1098/rspb.2012.1621.
- Ball, S. and Morris, R. 2015. *Britain's Hoverflies: A Field Guide* (second edition). Hampshire, UK. Princeton University Press.
- Bailey *et al.* 2014 Distance from forest edge affects bee pollinators in oilseed rape fields. *Ecology and Evolution* **4** (4), 370–80. doi:10.1002/ece3.924.
- Betts, M.G., Hadley A.S. and Kormann, U. 2019 The landscape ecology of pollination. *Landscape Ecology* **34**, 961–6. doi:10.1007/s10980-019-00845-4.
- Bianchi, F.J., Booij, C.J.H. and Tscharntke, T. 2006 Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings of the Royal Society B: Biological Sciences* **273** (1595), 1715–27. doi:10.1098/rspb.2006.3530.
- Biesmeijer *et al.* 2006 Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science* **313** (5785), 351–354. doi:10.1126/science.1127863.
- Bodin *et al.* 2006 The value of small size: Loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications* **16**, 440–51. doi:10.1890/1051-0761(2006)016[0440:tvoss]2.0.co;2.
- Boscolo *et al.* 2017 Positive responses of flower visiting bees to landscape heterogeneity depend on functional connectivity levels. *Perspectives in Ecology and Conservation* **15** (1), 18–24. doi:10.1016/j.pecon.2017.03.002.
- Brosi, B.J., Armsworth, P.R. and Daily, G.C. 2008 Optimal design of agricultural landscapes for pollination services. *Conservation Letters* **1** (1), 27–36. doi:10.1111/j.1755-263X.2008.00004.x.
- Bullock, C. and Hawe, J. 2014 *The Natural Capital Value of Native Woodland in Ireland. Woodlands of Ireland*. Rathfarnham Castle, Dublin. Seismograph House.
- Byrne, F. and Barco-Trillo, J. 2019 The effect of management practices on bumblebee densities in hedgerow and grassland habitats. *Basic and Applied Ecology* **35**, 28–33. doi:10.1016/j.baae.2018.11.004.
- Calabrese, J.M. and Fagan, W.F. 2004 A comparison-shopper's guide to connectivity metrics. *Frontiers in Ecology and the Environment* **2** (10), 529–36. doi:10.1890/1540-9295(2004)002[0529:ACGTCM]2.0.CO;2.
- Carrié *et al.* 2017 Relationships among ecological traits of wild bee communities along gradients of habitat amount and fragmentation. *Ecography* **40** (1), 85–97. doi:10.1111/ecog.02632.
- Carvalho *et al.* 2011 Natural and within-farmland biodiversity enhances crop productivity. *Ecology Letters* **14**, 251–9. doi:10.1111/j.1461-0248.2010.01579.x.
- Chagnon, M. 2008 *Causes and effects of the worldwide decline in pollinators and corrective measures*. Canadian Wildlife Federation. Quebec Regional Office, Canada.
- Connelly, H., Poveda, K. and Loeb, G. 2015 Landscape simplification decreases wild bee pollination services to strawberry. *Agriculture Ecosystems and Environment* **211**, 51–6. doi:10.1016/j.agee.2015.05.004.
- Cranmer, L., McCollin, D. and Ollerton, J. 2011 Landscape structure influences pollinator movements and directly affects plant reproductive success. *Oikos* **121** (4), 562–8. doi:10.1111/j.1600.
- Dainese *et al.* 2019 A global synthesis reveals biodiversity-mediated benefits for crop production. *Science Advances* **5** (10), eaax0121. doi:10.1126/sciadv.aax0121.
- Delattre, T., Vernon, P. and Burel, F. 2013 An agri-environmental scheme enhances butterfly dispersal in European agricultural landscapes. *Agriculture, Ecosystems and Environment* **166**, 102–9. doi:10.1016/j.agee.2011.06.018.
- Dormann *et al.* 2013 Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* **35**, 1–20. doi:10.1111/j.1600-0587.2012.07348.x.
- European Commission (EC) 2020a EU Biodiversity Strategy for 2030 – Bringing nature back into our lives. Communication from the commission to the European parliament, the council, the European economic and social committee and the committee of the regions. <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52020DC0380>. European Commission (EC) 2020b *Farm to Fork Strategy*. For a fair, healthy and environmentally-friendly food system. https://ec.europa.eu/food/farm2fork_en. 02 November 2020.
- European Environment Agency 2019 Climate change adaptation in the agriculture sector in Europe. Luxembourg, Publications Office of the European Union. <https://www.eea.europa.eu/publications/cc-adaptation-agriculture>. 03 November 2020.
- Eigenbrod, F. 2016 Redefining landscape structure for ecosystem services. *Current Landscape Ecology Reports* **1** (2), 80–6. doi:10.1007/s40823.
- Else, G.R. and Edwards, M. 2018 *Handbook of the Bees of the British Isles*. London, UK. Ray Society Monographs, The Ray Society, c/o The Natural History Museum.
- Economic and Social Research Institute (ESRI) 2016 ArcGIS Desktop: version 10.5. Redlands, CA. Environmental Systems Research Institute.
- Fahrig, L. 2017 Ecological responses to habitat fragmentation *per se*. *Annual Review of Ecology, Evolution, and Systematics* **48** (1), 1–23. doi:10.1146/annurev-ecolsys-110316.
- Falk, S. 2015 *Field guide to the bees of Great Britain and Ireland*. London, UK, and New York, USA. Bloomsbury.
- Farine, D.R. 2017 A guide to null models for animal social network analysis. *Methods in Ecology and Evolution* **8**, 1309–20. doi:10.1111/2041-210X.12772.
- Fossitt, J.A. 2000 *A Guide to Habitats in Ireland*. Kilkenny, Ireland. The Heritage Council.
- Fuller *et al.* 2017 Local-scale attributes determine the suitability of woodland creation sites for Diptera. *Journal of Applied Ecology* **55** (3), 1173–84. doi:10.1111/1365.
- Garibaldi *et al.* 2011 ., Stability of pollination services decreases with isolation from natural areas despite honey bee visits. *Ecology Letters* **14** (10), 1062–72. doi:10.1111/j.1461-0248.2011.01669.x.
- Garibaldi *et al.* 2013 Wild pollinators enhance fruit set of crops regardless of honey bee abundance. *Science* **339**, 1608–11. doi:10.1126/science.1230200.
- Gaytán, Á., Ricarte, A. and González-Bornay, G. 2020 Hoverfly diversity (Diptera: Syrphidae) of Pyrenean oak woodlands in Central-Western Spain: a preliminary study with conservation outcomes. *Journal of Insect Conservation* **24**, 163–73. doi:10.1007/s10841-019-00208-z.
- Gittings *et al.* 2006 The contribution of open spaces to the maintenance of hoverfly (Diptera, Syrphidae) biodiversity in Irish plantation forests. *Forest*

- Ecology and Management* **237**, 290–300. doi:10.1016/j.foreco.2006.09.052.
- Greenleaf *et al.* 2007 Bee foraging ranges and their relationship to body size. *Oecologia* **153**, 589–96. doi:10.1007/s00442–007.
- Guisan, A. and Zimmermann, N.E. 2000 Predictive habitat distribution models in ecology. *Ecological Modelling* **135**, 147–86. doi:10.1016/s0304–3800(00)00354–9.
- Hadley, A.S. and Betts, M.G. 2012 The effects of landscape fragmentation on pollination dynamics: absence of evidence not evidence of absence. *Biological Reviews* **87**, 526–44. doi:10.1111/j.1469–185X.2011.00205.x.
- Haenke *et al.* 2014 Landscape configuration of crops and hedgerows drives local syrphid fly abundance. *Journal of Applied Ecology* **51** (2), 505–13. doi:10.1111/1365.
- Hendrickx *et al.* 2007 How landscape structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes. *Journal of Applied Ecology* **44** (2), 340–51. doi:10.1111/j.1365–2664.2006.01270.x.
- Herrault *et al.* 2015 Combined effects of area, connectivity, history and structural heterogeneity of woodlands on the species richness of hoverflies (Diptera: Syrphidae). *Landscape Ecology* **31** (4), 877–93. doi:10.1007/s10980–015–0304.
- Hong, X. and Mitchell, R.J. 2007 Backward elimination model construction for regression and classification using leave-one-out criteria. *International Journal of Systems Science* **38** (2), 101–113. doi:10.1080/00207720601051463.
- Huais *et al.* 2020 Forest fragments influence pollination and yield of soybean crops in Chaco landscapes. *Basic and Applied Ecology* **48**, 61–72. doi:10.1016/j.baae.2020.09.003.
- Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBS) 2016 Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production. In Potts *et al.* (eds) *The Assessment Report on Pollinators, Pollination and Food Production*. Bonn, Germany. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services 36 pages..
- Irish Environmental Protection Agency 2018 Catchments data package. <https://gis.epa.ie/GetData/Download> 30 July 2018.
- Jauker *et al.* 2009 Pollinator dispersal in an agricultural matrix: opposing responses of wild bees and hoverflies to landscape structure and distance from main habitat. *Landscape Ecology* **24**, 547–55. doi:10.1007/s10980–009–9331–2.
- Jing, S., Kryger, P. and Boelt, B. 2021 Review of seed yield components and pollination conditions in red clover (*Trifolium pratense* L.) seed production. *Euphytica* **217**, 69. doi:10.1007/s10681–021–02793–0
- Joshi *et al.* 2016 Proximity to woodland and landscape structure drives pollinator visitation in apple orchard ecosystem. *Frontiers in Ecology and Evolution* **43** (8), 1–9. doi:10.3389/fevo.2016.00038.
- Jovičić *et al.* 2017 Influence of landscape structure and land use on Merodon and Cheilosia (Diptera: Syrphidae): contrasting responses of two genera. *Journal of Insect Conservation* **21** (1), 53–64. doi:10.1007/s10841–016–9951–1.
- Jung, M. 2013 LecoS – A QGIS plugin for automated landscape ecology analysis. *PeerJ Pre Prints* **1**, e116v2. doi:10.7287/peerj.preprints.116v2.
- Kearns, C.A., Inouye, D.W. and Waser, N.M. 1998 Endangered mutualisms: the conservation of plant-pollinator interactions. *Annual Review of Ecology, Evolution, and Systematics* **29**, 83–112. doi:10.1146/annurev.ecolsys.29.1.83.
- Kennedy *et al.* 2013 A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecology Letters* **16** (5), 584–99. doi:10.1111/ele.12082.
- Kleijn *et al.* 2006 Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters* **9** (3), 243–54. doi:10.1111/j.1461–0248.2005.00869.x.
- Klein *et al.* 2007 Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society B: Biological Sciences* **274** (1608), 303–13. doi:10.1098/rspb.2006.3721.
- Kovács-Hostyánszki *et al.* 2017 Ecological intensification to mitigate impacts of conventional intensive land-use on pollinators and pollination. *Ecology Letters* **20** (5), 673–89. doi:10.1111/ele.12762.
- Kremen *et al.* 2004 The area requirements of an ecosystem service: crop pollination by native bee communities in California. *Ecology Letters* **7** (11), 1109–11. doi:10.1111/j.1461–0248.2004.00662.x.
- Krewenka *et al.* 2011 Landscape elements as potential barriers and corridors for bees, wasps and parasitoids. *Biological Conservation* **144** (6), 1816–25. doi:10.1016/j.biocon.2011.03.014.
- Krishnan *et al.* 2020 *The Pollination Services of Forests – A Review of Forest and Landscape Interventions to Enhance Their Cross-sectoral Benefits*. Forestry Working Paper No. 15. Rome. The Food and Agriculture Organization (FAO) and Bioversity International. doi:10.4060/ca9433en.
- Lawson, D.A. and Rands, S.A. 2019 The effects of rainfall on plant-pollinator interactions. *Arthropod-Plant Interactions* **13**, 561–69. doi:10.1007/s11829–019–09686–z.
- Met Éireann 2020 Monthly data. <https://www.met.ie/climate/available-data/monthly-data> 09 September 2020.
- MEA (Millennium Ecosystem Assessment) 2005 *Ecosystems and human well-being*. Washington, D.C: Island Press.
- Mitchell, M.G.E., Bennett, E.M., and Gonzalez, A. 2013 Linking landscape connectivity and ecosystem service provision: current knowledge and research gaps. *Ecosystems* **16**, 894–908. doi:10.1007/s10021–013–9647–2.
- Moquet *et al.* 2018 Conservation of hoverflies (Diptera, Syrphidae) requires complementary resources at the landscape and local scales. *Insect Conservation and Diversity* **11**, 72–87. doi:10.1111/icad.12245
- Moreira *et al.* 2016 Are pan traps colours complementary to sample community of potential pollinator insects? *Journal of Insect Conservation* **20** (4), 583–96. doi:10.1007/s10841–016–9890–x.
- Newman *et al.* 2019 Scaling and complexity in landscape ecology. *Frontiers in Ecology and Evolution* **7** (293), 1–16. doi:10.3389/fevo.2019.00293.
- Öckinger, E. and Smith H.G. 2006 Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes. *Journal of Applied Ecology* **44** (1), 50–9. doi:10.1111/j.1365–2664.2006.01250.x.

- Ollerton, J., Winfree, R. and Tarrant, S. 2011 How many flowering plants are pollinated by animals? *Oikos* **120**, 321–6. doi:10.1111/j.1600-0706.2010.18644.x.
- Peyraud *et al.* 2010 Producing milk from grazing to reconcile economic and environmental performances. Proceeding of the 23rd General Meeting of the European Grassland Federation, Kiel, Germany, 29 August – 02 September 2010. *Grassland Science in Europe* **15**, 865–79.
- Pfeiffer *et al.* 2019 Woodland and floral richness boost bumble bee density in cranberry resource pulse landscapes. *Landscape Ecology* **34**, 979–96. doi:10.1007/s10980-019-00810-1.
- Popic, T.J., Davila, Y.C. and Wardle, G.M. 2013 Evaluation of common methods for sampling invertebrate pollinator assemblages: net sampling out-perform pan traps. *PLoS ONE* **8** (6), e66665. doi:10.1371/journal.pone.0066665.
- Popov *et al.* 2017 Phytophagous hoverflies (Diptera: Syrphidae) as indicators of changing landscapes. *Community Ecology* **18** (3), 287–94. doi:10.1556/168.2017.18.3.7.
- Potts *et al.* 2010 Global pollinator declines: trends, impacts and drivers. *Trends in Ecology and Evolution* **25** (6), 345–53. doi:10.1016/j.tree.2010.01.007.
- Potts *et al.* 2011 Stability of pollination services decreases with isolation from natural areas despite honey bee visits. *Ecology Letters* **14**, 1062–72. doi:10.1111/j.1461-0248.2011.01669.x.
- Potts *et al.* 2015 *Status and trends of European pollinators. Key findings of the STEP project.* Sofia, Bulgaria. Pensoft Publishers.
- Power *et al.* 2016 Organic farming and landscape factors affect abundance and richness of hoverflies (Diptera, Syrphidae) in grasslands. *Insect Conservation and Diversity* **9**, 244–53. doi:10.1111/icad.12163.
- R Development Core Team 2013 *R version 3.5.2: A Language and Environment for Statistical Computing.* Vienna, Austria. R Foundation for Statistical Computing.
- Rader *et al.* 2016 Non-bee insects are important contributors to global crop pollination. *PNAS* **113** (1), 146–51. doi:10.1073/pnas.1517092112.
- Read *et al.* 2018 Tropical bird species have less variable body sizes. *Biology Letters* **14**, 20170453. doi:10.1098/rsbl.2017.0453.
- Ricketts, T.H. 2001 The matrix matters: effective isolation in fragmented landscapes. *The American Naturalist* **158** (1), 87–99. doi:10.1086/320863.
- Ricketts, T., Williams, N.M. and Mayfield, M.M. 2006 Chapter 11: Connectivity and ecosystem services: crop pollination in agricultural landscapes. In K.R. Crooks and M. Sanjayan (eds), *Connectivity conservation Top of Form Bottom of Form*, 255–90, Cambridge, UK. Cambridge University Press. doi:10.1017/CBO9780511754821.012.
- Ricketts *et al.* 2008 Landscape effects on crop pollination services: are there general patterns? *Ecology Letters* **11**, 499–515. doi:10.1111/j.1461-0248.2008.01157.x.
- Riitters *et al.* 1995 A factor analysis of landscape pattern and structure metrics. *Landscape Ecology* **10** (1), 23–39. doi:10.1007/bf00158551.
- Roberts, H.P., King, D.I. and Milam, J. 2017 Factors affecting bee communities in forest openings and adjacent mature forest. *Forest Ecology and Management* **394**, 111–22. doi:10.1016/j.foreco.2017.03.027.
- Rotchés-Ribalta *et al.* 2021 Assessment of semi-natural habitats and landscape features on Irish farmland: new insights to inform EU common agricultural policy implementation. *Ambio* **50**, 346–59. doi:10.1007/s13280-020-01344-6.
- Saura, S. and Rubio, L. 2010 A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. *Ecography* **33**, 523–53. doi:10.1111/j.1600-0587.2009.05760.x.
- Schirmel *et al.* 2016 Trait composition and functional diversity of spiders and carabids in linear landscape elements. *Agriculture, Ecosystems and Environment* **235**, 318–28. doi:10.1016/j.agee.2016.10.028.
- Senapathi *et al.* 2017 Landscape impacts on pollinator communities in temperate systems: evidence and knowledge gaps. *Functional ecology* **31** (1) 26–37. doi:10.1111/1365-2435.12809.
- Slancarová *et al.* 2014 Does the surrounding landscape heterogeneity affect the butterflies of insular grassland reserves? A contrast between composition and configuration. *Journal of Insect Conservation* **18** (1), 1–12. doi:10.1007/s10841-013-9607-3.
- Smith *et al.* 2011 *Best Practice Guidance for Habitat Survey and Mapping.* Kilkenny, Ireland. The Heritage Council.
- Speight, M.C.D., Castella, E. and Sarthou, J.P. 2016 StN 2016. Syrph the Net on CD, Issue 11. In Speight *et al.* (eds) Dublin, Ireland. Syrph the Net Publications.
- Stanley, D.A., Gunning, D. and Stout, J.C. 2013 Pollinators and pollination of oilseed rape crops (*Brassica napus* L.) in Ireland: ecological and economic incentives for pollinator conservation. *Journal of Insect Conservation* **17**, 1181–9. doi:10.1007/s10841-013-9599-z.
- Steffan-Dewenter, I. and Tscharntke, T. 1999 Effects of habitat isolation on pollinator communities and seed set. *Oecologia* **121**, 432–40. doi:10.1007/s004420050949.
- Steffan-Dewenter *et al.* 2002 Scale-dependent effects of landscape context on three pollinator guilds. *Ecology* **83** (5), 1421–32. doi:10.1890/0012-9658(2002)083[1421:SD EOLC]2.0.CO;2.
- Stubbs, A. and Falk, S. 2002 *British Hoverflies: an Illustrated Identification Guide* (second edition). Reading, UK. British Entomological and Natural History Society.
- Sullivan *et al.* 2017 A national-scale model of linear features improves predictions of farmland biodiversity. *Journal of Applied Ecology* **54**, 1776–84. doi:10.1111/1365-2664.12912.
- Taki, H., Kevan, P.G. and Ascher, J.S. 2007 Landscape effects of forest loss in a pollination system. *Landscape Ecology*, **22** (10), 1575–87. doi:10.1007/s10980-007-9153-z.
- Taylor, Fahrig, L. and With, K.A. 2006 Chapter: 2. Landscape connectivity: a return to the basics. In K.R. Crooks and M. Sanjayan (eds), *Connectivity Conservation*, 29–43. Cambridge, UK. Cambridge University Press., doi:10.1017/CBO9780511754821.
- Thomas, R. and the Guidebook Team 2017 *Data Analysis with R Statistical Software.* A Guidebook for Scientists. UK. Eco-explore.
- Totland, O. 1994 Intraseasonal variation in pollination intensity and seed set in an alpine population of *Ranunculus acris* in southwestern Norway. *Ecography* **17**, 159–65. doi:10.1111/j.1600-0587.1994.tb00089.x.

- Valdés *et al.* 2019 High ecosystem service delivery potential of small woodlands in agricultural landscapes. *Journal of Applied Ecology* **57**, 4–16. doi:10.1111/1365-2664.13537.
- Van Geert, A., Van Rossum, F. and Triest, L. 2010 Do linear landscape elements in farmland act as biological corridors for pollen dispersal? *Journal of Ecology* **98** (1), 178–87. doi:10.1111/j.1365.
- Vanbergen, A.J. and the Insect Pollinators Initiative 2013 Threats to an ecosystem service: pressures on pollinators. *Frontiers in Ecology and the Environment* **11** (5), 251–9. doi:10.1890/120126.
- Venables, W.N. and Ripley, B.D. 2002 Modern Applied Statistics with S (fourth edition). New York, USA. Springer.
- Viana *et al.* 2012 How well do we understand landscape effects on pollinators and pollination services? *Journal of Pollination Ecology* **7** (5), 31–41. doi:10.26786/1920-7603(2012)2.
- Walz, U. 2011 Landscape structure, landscape metrics and biodiversity. *Living Reviews in Landscape Research* **5** (3), 1–16. <http://www.livingreviews.org/lrlr-2011-3>.
- Willaert, E. 2019 Wild pollinators in agricultural landscapes. The impact of field margins at a small spatial scale. Unpublished MSc thesis, Ghent University, Belgium.
- Winfree *et al.* 2007 Native bees provide insurance against ongoing honey bee losses. *Ecology Letters* **10**, 1105–13. doi:10.1111/j.1461-0248.2007.01110.x.
- Winfree *et al.* 2008 Wild bee pollinators provide the majority of crop visitation across land-use gradients in New Jersey and Pennsylvania, USA. *Journal of Applied Ecology* **45**, 793–802. doi:10.1111/j.1365-2664.2007.01418.x
- Woodcock *et al.* 2019 Meta-analysis reveals that pollinator functional diversity and abundance enhance crop pollination and yield. *Nature Communications* **10**, 1481. doi:10.1038/s41467-019-09393-6.
- Wright, I.R., Roberts, S.P.M. and Collins, B.E. 2015 Evidence of forage distance limitations for small bees (Hymenoptera: Apidae). *European Journal of Entomology* **112** (2), 303–10. doi:10.14411/eje.2015.028.
- Yachi, S. and Loreau, M. 1999 Biodiversity and ecosystem productivity in a fluctuating environment: the insurance hypothesis. *PNAS USA* **96**, 1463–8. doi:10.1073/pnas.96.4.1463.
- Zurbuchen, A. 2010 Distance matters. Impact of increasing foraging distances on population dynamics in native bees. Unpublished doctoral thesis, ETH Zürich, Switzerland. doi:10.3929/ethz-a-006020778.

DOES LANDSCAPE STRUCTURE AFFECT THE PRESENCE OF WOODLAND SPECIALIST POLLINATORS IN FARMLAND? IMPLICATIONS FOR AGRI-ENVIRONMENT SCHEME DESIGN

Table A1—Average temperature and rainfall registered per month by the closest meteorological station (Met Éireann, 09 September 2020, URL: <https://www.met.ie/climate/available-data/monthly-data>).

	<i>Precipitation (mm)</i>				<i>Air Temperature (°C)</i>			
	<i>May</i>	<i>Jun</i>	<i>Jul</i>	<i>Aug</i>	<i>May</i>	<i>Jun</i>	<i>Jul</i>	<i>Aug</i>
Co. Sligo	78.9	75.5	110.9	72.9	10.1	12.5	15.4	14.9
Co. Wexford	101.9	26.5	81.8	35.1	11.1	12.8	16.0	15.7

Table A2—Compositional and configurational metrics calculated for all sampled parcels.

Compositional metrics	
<i>Total area of optimal habitat</i>	Sum of all the individual habitat patches listed in Table 1 present on the buffer of 500m (see Figure 2 C in the main text for example). Only the areas inside the buffer were included.
<i>Area of intensive production</i>	Total area of intensive land-use dedicated to food or fibre production (including improved grasslands, tillage fields, other crop fields and conifer plantation— following Fossitt (2000) classification.
<i>Area of semi-natural pastures</i>	Total area of semi-natural pastures, which included the area of semi-natural grasslands, heathland, peatlands and areas where a combination of these three habitats is present (e.g. transition habitats) (following Fossitt 2000 and O'Neill <i>et al.</i> 2013)
<i>Area of semi-natural grasslands</i>	Total area of semi-natural grasslands – area of semi-natural grasslands was considered separately and relates to the ecological integrity of the grassland as a result of management (livestock units, nutrient inputs, reseeded) and follows Fossitt (2000), Sullivan <i>et al.</i> (2010), Devaney <i>et al.</i> (2013) and O'Neill <i>et al.</i> (2013).
<i>Total length of hedgerows</i>	Total length of hedgerows (woody linear feature). Hedges have been shown to be valuable landscape elements for pollinators by providing cavity nesting for bees and wasps, floral resources (references) and by acting as corridors (e.g. Hannon and Sisk 2009; Krewenka <i>et al.</i> 2011, Garratt <i>et al.</i> 2017; Volpato <i>et al.</i> 2019).
<i>Area of the closest patch of optimal habitat</i>	The area of the closest optimal habitat (any habitat patch from the list present in Table 1) from the pan traps.
<i>Greatest patch area</i>	The area of the greatest patch of 'optimal habitat'.
<i>Edge length</i>	Total edge length was calculated for all the existing patches of 'optimal habitat' and equals the total length of all patches from this specific class (determined using the LECOS (Jung, 2013) extension for QGIS - QGIS Development Team 2013).

Edge density Edge density was determined using the LECOS extension for QGIS (edge length divided by number of forest patches) (Jung 2013).

Configuration metrics

Distance from the trap to optimal habitat Distance from pan trap to the closest ‘optimal habitat’ area (shortest path) was determined in ArcGIS.10.5.

Connectivity of linear feature to optimal habitat Binary variable: if the linear feature was physically linked/connected to an area of ‘optimal habitat’, it was coded as ‘yes’ and if not it was coded as ‘no’. We considered that the linear features where the traps were placed was connected to an ‘optimal habitat’ patch if connected directly or connected to another linear feature that is itself connected to an ‘optimal habitat’.

Production fields average size This variable was calculated as the average size of the areas/fields dedicated to food or fibre production that were present in the 500m buffers.

Splitting Index S (SPLI) and Landscape Division Index (LDI) These two metrics measure the degree of fragmentation of a landscape and are based on the ability of two animals – placed in different areas somewhere in a region – to find each other within the landscape (see Jaeger (2000) for full description and formulas). Both metrics were calculated through LECOS extension for QGIS.

Patch Cohesion Index This index measures the physical connectedness of patches of ‘optimal habitats’ and increases as the patch type becomes more aggregated in its distribution (more physically connected). Because is less sensitive to small changes in patch perimeters it is considered as a superior predictor of dispersal success in realistic landscape (see Schumaker, 1996). Again, the value of this index was obtained from the LECOS extension for QGIS.; Saura and Rubio 2010; Baranyi *et al.* 2011).

Functional connectivity metrics Using Connefor 2.6. (Saura and Torné 2009; <http://www.conefor.org/>) we extracted seven (*dPA*, *dF*, *dFWA*, *dPC*, *dpCcintra*, *dpCflux*, *dPCcon* – Urban and Keit, 2001; Saura and Rubio 2010; Baranyi *et al.* 2011) variables that provide information on the degree of connectedness or isolation of the suitable patch that is closer to the traps. The value of each variable per patch is calculated in relation to the whole the landscape (i.e. sub-catchment) (see Figure 2 A and B). The first step to obtain these indices was to generate the connectors and nodes files using the ArcGIS plug-in (Jenness 2011) by defining a maximum theoretical value of dispersion between patches of 600m (maximum distance for the selected species to be moving in the landscape from their nesting sites). These two files were then inputted in Connefor and the indexes were determined using three different scenarios of average mobility of the species: a) very low mobility (50m); b) intermediate levels of mobility (250m); and c) high mobility (500m). The probability was set to 0.1 for the three scenarios. Thus, for each single patch in the whole landscape, the 7 connectivity indices varied according to the distances. We only extracted and considered the values of each index (for each distance – 50, 250m, 500m) for the closest patch of optimal habitat to assess the role of connectivity in the abundance of the selected pollinators. In total 21 connectivity metrics were extracted for each sampled point.

Metrics explanation:

dA = Area dependent index

dF = Index of dispersal flux (F)

dFWA (Area-weighted flux) = Sum of the products of the direct dispersal probability between each pair of nodes and the attributes (*dA*) of those nodes;

dPC = *dpCcintra* + *dpCflux* + *dPCcon*;

dpCcintra = Intrapatch connectivity index;

dpCflux = Index that provides information on how well connected a patch is to others in the landscape.

dPCcon = Index related to how important the patch is for the other patches to be connected.

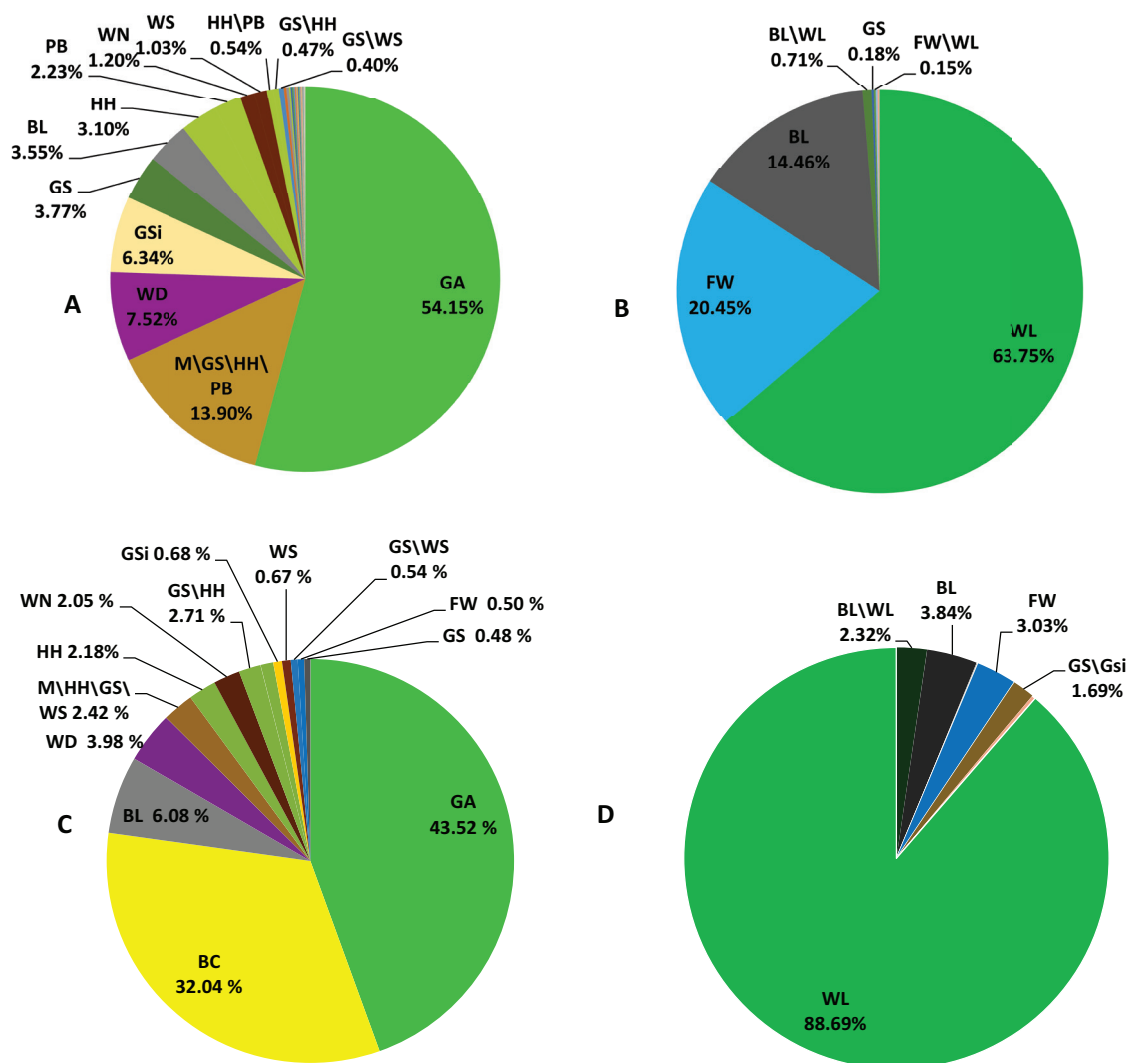


Figure A1—Proportion of the most common terrestrial habitats in the study regions: Pie charts A and B correspond to the proportion of habitats in the Co. Sligo sub-catchment (areal habitats and linear habitats, respectively); Pie charts C and D correspond to the proportion of habitats in the Co. Wexford sub-catchment (areal habitats and linear habitats respectively); only areal habitats that occupy more than 0.40% of the total area are shown; likewise, only linear habitats in that occupy represent more than 0.17% of the total length of linear features are shown (see Table A3 for habitat codes).

Table A3—Level 2 of Fossitt (2000) habitats code with corresponding description (used for mapping all habitats in both sub-catchments).

Code	Description	Code	Description
BC	Cultivated land	HH	Heathlands
BL	Built land (includes earthbanks and stonewalls)	LS	Littoral sediment
ED	Disturbed ground	MW	Marine water body
FL	Lakes and ponds	PB	Peatlands
FW	Watercourses	SR	Sub-littoral rock
GA	Improved grassland	WD	Highly modified/non-native forest
GS	Semi-natural grassland	WL	Linear woodland/scrub
GSi	Semi-improved grassland	WN	Semi-natural woodland
HD	Dense bracken	WS	Scrub/transitional woodland

Table A4—Results of Kruskal-Wallis tests for comparison of means per region (Co. Wexford vs Co. Sligo), for explanatory variables introduced in the model. Significance levels: ‘*’ 0.001 ‘**’ 0.01 ‘*’ 0.05.**

<i>Explanatory variable</i>	<i>K-W test results</i>
<i>Total area of optimal habitat (in 500m buffer)</i>	Chi-squared = 0.256, df = 1, p-value = 0.613
<i>Area of semi-natural grasslands (in 500m buffer)</i>	Chi-squared = 4.510, df = 1, p-value = 0.030*
<i>Area of semi-natural pastures (in 500m buffer)</i>	Chi-squared = 1.032, df = 1, p-value = 0.310
<i>Area of the closest patch of optimal habitat</i>	Chi-squared = 0.008, df = 1, p-value = 0.927
<i>Distance from the trap to optimal habitat</i>	Chi-squared = 0.931, df = 1, p-value = 0.335
<i>dPCcon500</i>	Chi-squared = 0.078, df = 1, p-value = 0.780
<i>dF500</i>	Chi-squared = 1.540, df = 1, p-value = 0.215
<i>Production fields average size (in 500m buffer)</i>	Chi-squared = 1.910, df = 1, p-value = 0.168
<i>Linear feature connected to optimal habitat</i>	Chi-squared = 1.246, df = 1, p-value = 0.264
<i>Total length of hedgerows</i>	Chi-squared = 0.762, df = 1, p-value = 0.382

REFERENCES

- Baranyi *et al.* 2011 Contribution of habitat patches to network connectivity: redundancy and uniqueness of topological indices. *Ecological Indicators* **11**, 1301–10. doi:10.1016/j.ecolind.2011.02.003.
- Devaney *et al.* 2013 *Irish Semi-natural Grasslands Survey Annual Report No. 4: Western Seaboard Counties (Clare, Galway, Kerry, Limerick, Mayo) County Tipperary*. Dublin, Ireland. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht. 68pp.
- Fossitt, J.A. 2000 *A Guide to Habitats in Ireland*. Kilkenny, Ireland. The Heritage Council.
- Garratt *et al.* 2017 The benefits of hedgerows for pollinators and natural enemies depends on hedge quality and landscape context. *Agriculture, Ecosystems and Environment* **247**, 363–70.
- Hannon, L.E. and Sisk, T.D. 2009 Hedgerows in an agri-natural landscape: potential habitat value for native bees. *Biological Conservation* **142** (10), 2140I.
- Jaeger, J.A.G. 2000 Landscape division, splitting index, and effective mesh size: new measures of landscape fragmentation. *Landscape Ecology* **15** (2), 115–30.
- Jenness, J. 2011 Conefor inputs tool for ArcGIS 10.x. Revision 1.0.218, April 9, 2016. http://www.jennessent.com/arcgis/conefor_inputs.htm
- Jung, M. 2013 LecoS - A QGIS plugin for automated landscape ecology analysis. *Peer J PrePrints* 1:e116v2 <https://doi.org/10.7287/peerj.preprints.116v2>.
- Krewenka *et al.* 2011 Landscape elements as potential barriers and corridors for bees, wasps and parasitoids. *Biological Conservation* **144** (6), 1816–25.
- O’Neill *et al.* 2013 *The Irish Semi-Natural Grasslands Survey 2007–2012. Irish Wildlife Manuals, No. 78*. Dublin. National Parks and Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Ireland.
- QGIS Development Team 2013 QGIS Geographic Information System. Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>
- Saura, S. and Rubio, L. 2010 A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. *Ecography* **33**, 523–37.
- Saura, S. and Torné, J. 2009 Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. *Environmental Modelling and Software* **24**, 135–9.
- Schumaker, N.H. 1996 Using landscape indices to predict habitat connectivity. *Ecology* **77**, 1210–25.
- Sullivan *et al.* 2010 The ecological status of grasslands on lowland farmlands in western Ireland and implications for grassland classification and nature value assessment. *Biological Conservation* **143** (6), 1529–39.
- Urban, D. and Keitt, T. 2001 Landscape connectivity: a graph-theoretic perspective. *Ecology* **82** (5), 1205–18. doi:10.1890/0012-9658(2001)082[1205:lcagt-p]2.0.co;2.
- Volpato *et al.* 2019 Using Malaise traps to assess aculeate Hymenoptera associated with farmland linear habitats across a range of farming intensities. *Insect Conservation and Diversity* **13**, 229–38.