RESEARCH ARTICLE

Impacts of management at a local and landscape scale on pollinators in semi-natural grasslands

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Abstract

1. Managing farmland to benefit biodiversity is becoming increasingly necessary to combat biodiversity declines and maintain ecosystem services. Results-based agri-environmental schemes are a tool used to achieve this by paying farmers based on environmental results delivered. These schemes often utilise plant indicator species to assess results at field scale; however, it is unknown if focusing on enhancing a subset of one biodiversity group within results-based schemes impacts wider biodiversity, and whether local-scale implementation of results-based schemes or/and the wider landscape are more important drivers of biodiversity patterns.

2. Insect pollinators provide important pollination services for many crop and wild plants, and as mobile organisms often experience landscape at large spatial scales. We tested whether insect pollinators are affected at local scale by a results-based scheme scored based on plant indicators, or if landscape management is more important, and whether there were different responses between taxon-specific groups. Bumblebees, solitary bees, hoverflies and butterflies were sampled using transects and pan traps in 23 fields with varying scores assigned by the scheme, situated in high-intensity (≥65% improved grassland) or low-intensity (≥65% semi-natural grassland) landscapes.

3. Results indicate taxon-specific responses to local and landscape management in semi-natural grasslands. Bumblebees responded positively to local-scale management in fields with higher floral diversity, whereas hoverflies and butterflies responded positively to low-intensity landscape management.

4. *Synthesis and applications*. Using plant species as indicators for biodiversity in agri-environment schemes can have indirect benefits for non-target taxa like bumblebees, but broader indicators should be developed to incorporate other pollinator groups. Pollinator groups respond differently to local and landscape management in semi-natural grasslands. Agri-environmental management should consider a range of different management measures and landscape scale approaches where possible, to maximise benefits for a range of pollinator taxa.

1 | INTRODUCTION

Insect pollinators provide an important ecosystem service to crops and wild plant species globally (Klein et al., 2007; Ollerton et al., 2011); a service estimated to contribute $235–$577 billion annually to agriculture alone (IPBES, 2016). However, insect pollinators have experienced global declines with land intensification highlighted as a leading factor (Potts et al., 2010).

Approximately 40% of global terrestrial land surface is used for agriculture (Ramankutty et al., 2008), and thus, pollinator conservation must occur in these areas with the added benefit of delivery of crop pollination services. Movement from extensive to intensive land management to increase agricultural output has replaced natural habitats, which provide pollinator forage and nesting resources, with species-poor systems (Fitzpatrick et al., 2007; Kennedy et al., 2013; Robinson & Sutherland, 2002). Agri-environmental schemes (AESs) are used in the EU and elsewhere to incentivise farmers to conserve farmland biodiversity (Batáry et al., 2015). However, research into the effectiveness of AESs reveals mixed results (Batary et al., 2011; Feehan et al., 2005; Kleijn et al., 2006). Many AESs are prescription based where payments are based on compliance with measures irrespective of outcome (Keenleyside et al., 2014) and so many do not achieve their desired outcome of biodiversity enhancement (Feehan et al., 2005; Roth et al., 2008). In addition, many measures employ broad-scale approaches which may influence their effectiveness between regions as they do not adapt to local conditions (Batáry et al., 2015).

In response to the mixed effects of AESs, results-based agri-environmental payment schemes (RBAPSs) have been developed where farmers are paid based on environmental results delivered rather than complying with measures. RBAPSs combine local knowledge with a strong evidence base, allow flexibility in land management and are area/region specific (O'Rourke & Finn, 2020). RBAPS measures are tailored to conserve specific species/groups, ecosystem services or habitats (Ekroos et al., 2014), ensuring clear and easy to implement objectives and easier farming community engagement. RBAPSs have being piloted across the EU to great success (Chaplin et al., 2019; McLoughlin, 2018; Page et al., 2019).

A challenge of RBAPSs is how to quantify biodiversity benefits. RBAPSs generally use indicator species to describe biodiversity, with plants commonly used as indicators. Promoting diverse plant communities could positively influence pollinator communities through provisioning of appropriate food resources (Carvell, 2002; Carvell et al., 2007) but whether plants act as useful indicators of diversity for other groups, for example, pollinators, within RBAPSs is unknown. In addition, pollinator taxa exhibit differing biology; therefore, taxon-specific responses to enhancement of specific plant species within an RBAPS may exist.

Due to logistical and administrative challenges of landscape-wide management, AESs including RBAPSs are largely implemented at field scale, with payments linked to individual fields. While solitary bees generally travel a couple of hundred metres from their nests, other pollinators like bumblebees can travel a few kilometres, therefore experience the landscape at larger spatial scales (Power et al., 2016; Redhead et al., 2016; Steffan-Dewenter et al., 2002). Therefore, it may be expected that responses to field and landscape AES management could vary between pollinator groups; for example, Rundlöf, Bengtsson et al. (2008) found that landscapes with high proportions of organic farming benefited butterfly richness irrespective of local management. However, additive and interactive influences of local-scale RBAPSs and surrounding landscape management are unknown.

Results-based agri-environmental payment schemes are growing in popularity, with proposals for their inclusion in wider agri-environmental policy (Herzon et al., 2018; O'Rourke & Finn, 2020). However, it is not known (a) how they influence wider biodiversity, (b) whether different taxa respond differently and (c) whether biodiversity patterns are influenced by management at local or/and landscape scales. We investigate whether non-target pollinator groups benefit from an RBAPS that uses plants as indicators of habitat quality in semi-natural grassland (SNG) systems at the local scale, and whether this was influenced by composition of the surrounding landscape. Understanding these relationships is key to pollinator conservation and the development of RBAPSs on a European and international scale moving forwards.

2 | MATERIALS AND METHODS

2.1 | Study region and site selection

This study was conducted in the Burren region in the West of Ireland (53.007°N 9.002°W); a biodiverse karst landscape used primarily for cattle grazing. Main habitats in the region include calcareous grassland, improved grassland and limestone pavement (Parr et al., 2009) with thin, free-draining rendzina soils, suitable for rough grazing (Finch et al., 1971) and some areas of brown earth. The Burren Programme is an award-winning RBAPS developed for the region as the one-size-fits-all approach of a national prescription-based AES was not effective in this unique landscape with localised farming practices (Dunford, 2001). This scheme assigns a conservation value to a field based on the presence of key plant species indicating a biodiverse meadow (Table S1; Text S1). Fields are scored on a A–D scale where A supports high numbers of positive indicator species typical of a biodiverse meadow, while D scores represent improved grassland (no positive indicator species; Figure S1a,b).

Study sites were selected to represent the range of conservation values in different landscapes. On examination of the habitats surrounding candidate sites, the main gradient identified was from intensive to extensive grasslands. A categorical approach was used to select sites in one of two landscapes at opposite ends of the gradient defined as low intensity (≥65% of SNG within a 2 km radius) or high intensity (≥65% of improved grassland within a 2 km radius; Text S2 for defining landscape context). The 2 km radius was chosen to represent a reasonable pollinator foraging range (Osborne et al., 2008; Zurbuchen et al., 2010) and is widely used in landscape scale studies (Ballare et al., 2019; Cohen et al., 2021). Within each landscape category, we selected fields (henceforth 'sites') that represented one of four conservation value scores (A–D) resulting in 23 sites with 12 sites in low-intensity and 11 in high-intensity landscapes (Figure 1; Table S2). To ensure independence of pollinator communities' sites were at least 1 km apart from each other. Although some pollinators are known to travel a few kilometres (Osborne et al., 2007), the Burren is florally diverse (Webb & Scannell, 1983) and pollinators are known to forage smaller distances when resources are common (Redhead et al., 2016).

## 2.2 | Pollinator and floral unit sampling

Bumblebees, solitary bees, hoverflies and butterflies were sampled using standardised transects and pan traps (Westphal et al., 2008). A 350-m transect was walked at each site three times from late-May to mid-September 2017. Routes were at least 2 m from field boundaries to avoid edge effects. Transects were walked between 9:00 and 18:00 at a slow steady pace for 50–60 min (total observation time per site between 170 and 180 min) in dry, bright conditions (temperature >13°C, wind force 0–4 Beaufort scale). Individuals were recorded up to 2.5 m on either side and in front of observer, and their behaviour recorded (e.g. flying). If an individual was observed visiting a plant, the plant species was recorded. Any individuals that could not be identified were captured using a sweep net and frozen for lab identification.

Pan trapping occurred twice in each site, in mid-June and late-August 2017. Sampling early in the season was avoided to prevent lethal sampling of bumblebee queens and solitary bees during their nest establishment phase. Plastic 12oz bowls (Musgrave Marketplace) were painted with UV-bright fluorescent paint (UV gear, UK). Nine traps were deployed in each field, in clusters of three (one blue, one yellow, one white), 20 m apart for 48 hr. Clusters were attached to 1 m wooden stakes using metal clamps to ensure traps were at the same height as vegetation to avoid attracting pollinators from outside the site (Figure S1c; Stanley & Stout, 2013). Traps were filled with water and a drop of ECOVER detergent to reduce surface tension. Collected specimens were stored in whirlpaks containing 70% IMS with 5% glycerol, before sorting and identification of wild bees and hoverflies in the laboratory using standard texts (Else, 2018; Stubbs & Falk, 2002). Three butterfly specimens were collected, but were not included in analyses, as this is not a standard butterfly sampling method.

Abundance and species richness of plants (excluding monocots) providing forage for insects during the sampling season were recorded using floral unit surveys. Eight 2 × 2 m quadrats were set 50 m apart along transects with one quadrat placed in the middle 25 m from the adjacent quadrats (9 quadrats per transect). All flowering plant species in bloom at the time of each survey were recorded, using Parnell et al. (2012), and the floral unit abundance of each counted (Dicks et al., 2002).

FIGURE 1 Burren region showing the 23 sites. Shapes represent conservation value scores assigned to each field within the Burren Programme (A; high numbers of positive plant indicators, D; no positive indicators). Sites in high-intensity landscapes (≥65% improved grassland) are in grey while sites in low-intensity (≥65% SNG) are in black. Map in top left corner shows Burren location in Ireland. Burren boundary is used with permission from BurrenLIFE and HNVS [High Nature Value Services], County Clare, retrieved April 2017
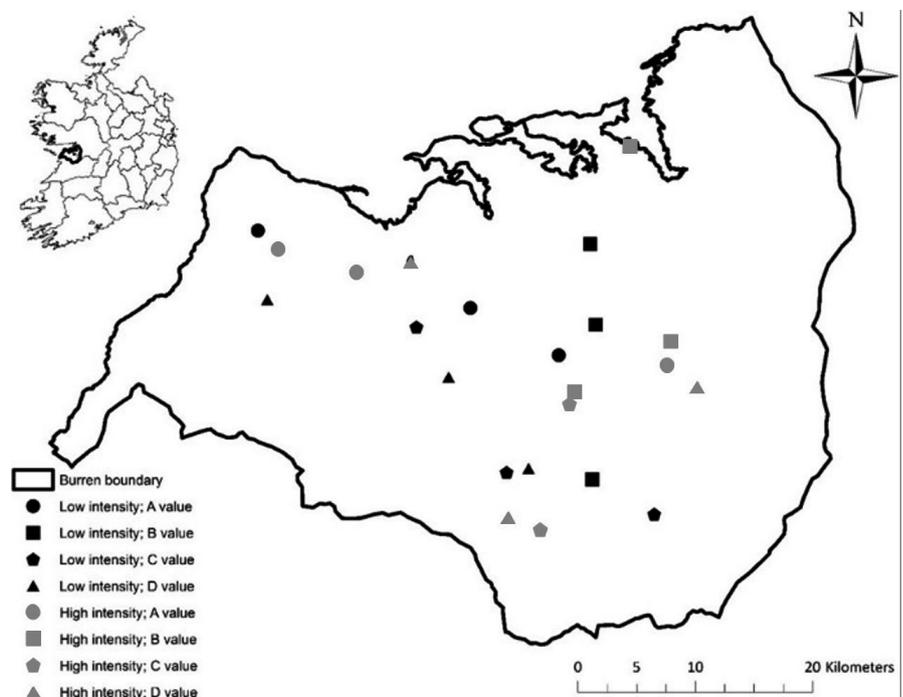
2.3 | Data analysis

Statistical analyses were conducted in R version 3.6.0 (R Core Team, 2019). Transect and pan trap data were analysed separately to account for differences in sampling techniques. Transect data included all individuals observed regardless of behaviour. Shannon diversity of transect data was calculated for pollinator groups for each site using the ᴠᴇɢᴀɴ package (Oksanen et al., 2019). Some pan traps were lost or damaged and to account for differences in sampling effort between sites, richness and Shannon diversity estimates for each species group were calculated using the package ɪNEXT (Hsieh et al., 2019). One site was omitted from analysis as 10 of 18 pan traps were damaged. To standardise pan trap abundance data, abundance was divided by number of undamaged traps at a site and multiplied by total number of traps placed in that site across both sampling periods (18).

To analyse relationships between landscape intensity, conservation value and floral richness and abundance (fixed effects) and abundance, richness and diversity of each pollinator group (response variables), generalised linear models (GLMs) and linear models (LMs) were used. Relationships were detected between conservation value and floral richness, and between floral richness and floral abundance (Figure S3); therefore, floral richness and abundance were analysed individually in separate models. Response variables were pooled at site level across sampling periods, as sample sizes at each period were small for some groups. LMs were used for non-count data, and Poisson error distribution using a log-link function was specified for count data GLMs with negative binomial used in cases of overdispersion. Where needed to improve model fit, data were log-transformed and analysed using LM. Models including conservation value and landscape intensity started with an interaction between fixed effects. The best-fitting model was selected by sequentially removing interactions and terms, and checking whether models differed from each other and from the null model using Chi-square tests (Zuur, 2009). Models were validated by inspecting residuals and Q-Q plots. The amount of variation each model explained was determined using explained deviance for GLMs and adjusted R^2 for LMs (Zuur, 2009). Post hoc Tukey's test was used to determine differences between levels of fixed effects using the package ᴍᴜʟᴛᴄᴏᴍᴘ (Hothorn et al., 2008). One outlier was detected for hoverfly abundance across both sampling methods, so GLMs were run with and without outliers. Model validation results did not change when outliers were removed and since outliers represented genuine samples, they were kept in the models.

To identify relationships between community composition of each species group with landscape intensity and conservation values, non-metric multidimensional scaling (NMDS) was used (McCune, 2002) in the ᴠᴇɢᴀɴ package (Oksanen et al., 2019). Community matrices were created of abundance of each species for each site across all sampling periods, for each taxonomic group (bumblebees, solitary bees, hoverflies, butterflies and floral units) and sampling method. Single counts and sites with single/no counts were removed as these would skew the main patterns within the dataset (Table S3). Data were analysed using Shepard's plots to determine if original dissimilarity would be preserved in NMDSs reduced number of dimensions. Analysis of similarities (ANOSIM) using Bray–Curtis index was used to test differences between explanatory variables with each matrix.

3 | RESULTS

In total, 872 individuals were observed on transects representing 377 bumblebees (11 species), 46 solitary bees (4 species), 135 butterflies (17 species) and 314 hoverflies (32 species). Two hundred and forty six individuals were collected from pan traps; 53 bumblebees (6 species), 41 solitary bees (6 species) and 152 hoverflies (21 species; Tables S2 and S4, and Text S3 for discussion on sampling method effectiveness). Overall, 80 flowering plant species were recorded across 24 families (24,741 floral units, Table S4), of which 46 species were observed to receive visits from insects (591 visits). Plants most frequently visited by insects were *Trifolium repens* (71), *Ranunculus repens* (69) and *Centaurea nigra* (68).

Floral community composition differed at local scale among conservation values ($R = 0.601$, $p = 0.001$), but not between landscape intensities ($R = 0.009$, $p = 0.35$; Figure S2).

3.1 | Bumblebees

Using transect data, bumblebee community composition varied at local scale between conservation values A and D ($R = 0.201$, $p = 0.02$) but not between landscape intensities ($R = 0.114$, $p = 0.08$; Figure S4a,b). Bumblebee abundance and richness were related to conservation value (explaining 39% and 54% variation respectively) while diversity was influenced by conservation value and landscape intensity (explaining 67% variance; Table 1). D fields had lower richness (Figure 2a), abundance and diversity compared to A and B fields, while diversity decreased from low- to high-intensity landscapes (Figure S5). Bumblebee abundance, richness and diversity were positively influenced by fields with higher floral richness (explaining 21%, 40% and 45% variation respectively) and abundance (explaining 52%, 52% and 27% variation respectively; Table S5).

Pan trap data revealed no differences between bumblebee community composition at local ($R = 0.01$, $p = 0.45$) or landscape level ($R = -0.076$, $p = 0.70$; Figure S4c,d). Relationships between bumblebee richness, abundance and diversity were explained by null models (Figure S6; Table S5).

3.2 | Hoverflies

Using transect data, hoverfly community composition did not differ at local scale ($R = -0.032$, $p = 0.66$) or landscape intensities ($R = 0.084$, $p = 0.12$; Figure S7a,b). Hoverfly abundance, richness and diversity were influenced by landscape intensity (17%, 38%

TABLE 1 Estimates, standard errors (*SE*) and 95% confidence intervals (95% CI) of fixed effects included in best-fitting model explaining variation in bumblebee richness, abundance and diversity sampled using transects. Estimates and CI of richness and abundance GLM were backtransformed to account for log-link function; therefore, any CIs not including 1 (null value) are in bold. Diversity LM did not require transformation; therefore, any CIs not including 0 (null value) are in bold

Response	Fixed effect(s)	Estimate	*SE*	95% CI Lower limit	Upper limit
Richness	Conservation value A (Intercept)	**5.00**	0.18	**3.42**	**7.01**
	Conservation value B	1.07	0.25	0.65	1.76
	Conservation value C	0.80	0.29	0.45	1.40
	Conservation value D	**0.40**	0.34	**0.20**	**0.76**
Abundance	Conservation value A (Intercept)	**15.83**	0.17	**11.50**	**21.98**
	Conservation value B	**1.57**	0.22	**1.01**	**2.44**
	Conservation value C	0.92	0.25	0.57	1.50
	Conservation value D	0.63	0.25	0.39	1.02
Diversity	Conservation value A + high-intensity landscape (Intercept)	**1.16**	0.13	**0.88**	**1.44**
	Conservation value B	−0.14	0.17	−0.50	0.21
	Conservation value C	−0.45	0.18	−0.82	−0.083
	Conservation value D	**−0.93**	0.17	**−1.29**	**−0.58**
	Low-intensity landscape	**0.41**	0.12	**0.15**	**0.66**

FIGURE 2 (a) Mean bumblebee (±*SE*) and (c) mean hoverfly (±*SE*) species richness for each conservation value. Mean (b) bumblebee (±*SE*) and (d) hoverfly (±*SE*) in species richness high- and low-intensity landscapes. Letters indicate significant differences. Data collected using transects
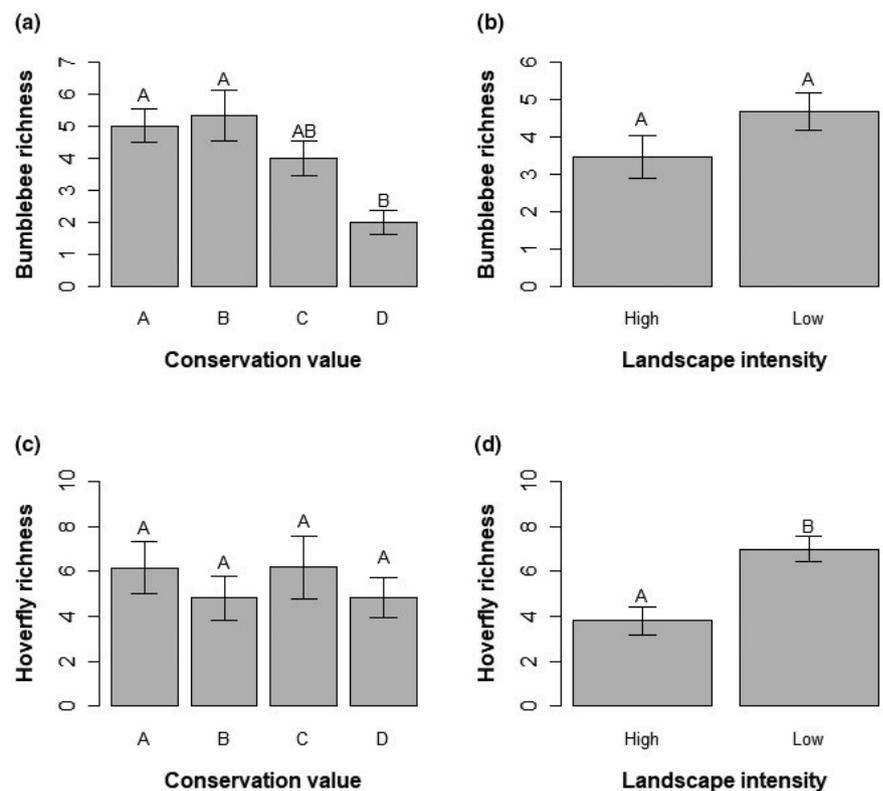
and 28% respectively; Table 2) but not at local scale (Figure 2c,d; Figure S8). All three measures decreased when moving from low- to high-intensity landscapes. Hoverfly abundance was positively related to floral abundance (explaining 15% variation), whereas hoverfly richness and diversity were not influenced by floral richness or abundance (Table S7).

Pan trap data showed no difference in hoverfly community composition between landscape intensities (R = 0.016, p = 0.37) or local scale (R = 0.081, p = 0.22; Figure S7c,d). Hoverfly richness was influenced by the interaction between landscape intensity and conservation value where sites with a C score in low-intensity landscapes had higher richness (explaining 49% variation). This pattern was driven by no hoverflies being

TABLE 2 Estimates, standard errors (*SE*) and 95% confidence intervals (95% CI) of fixed effects of best-fit models explaining variation in hoverfly richness, abundance and diversity sampled using transects. Estimates and CI of richness and abundance GLM were backtransformed to account for log-link function; therefore, any CIs not including 1 (null value) are in bold. Diversity LM did not require transformation; therefore, any CIs not including 0 (null value) are in bold

|				95% CI	
Response	Fixed effect(s)	Estimate	SE	Lower limit	Upper limit
Hoverfly (transect)					
Richness	High-intensity landscape (Intercept)	3.82	0.15	2.78	5.09
	Low-intensity landscape	1.83	0.19	1.27	2.68
Abundance	High-intensity landscape (Intercept)	8.73	0.24	5.60	14.23
	Low-intensity landscape	2.08	0.32	1.11	3.90
Diversity	High-intensity landscape (Intercept)	1.05	0.14	0.76	1.34
	Low-intensity landscape	0.60	0.19	0.19	1.00
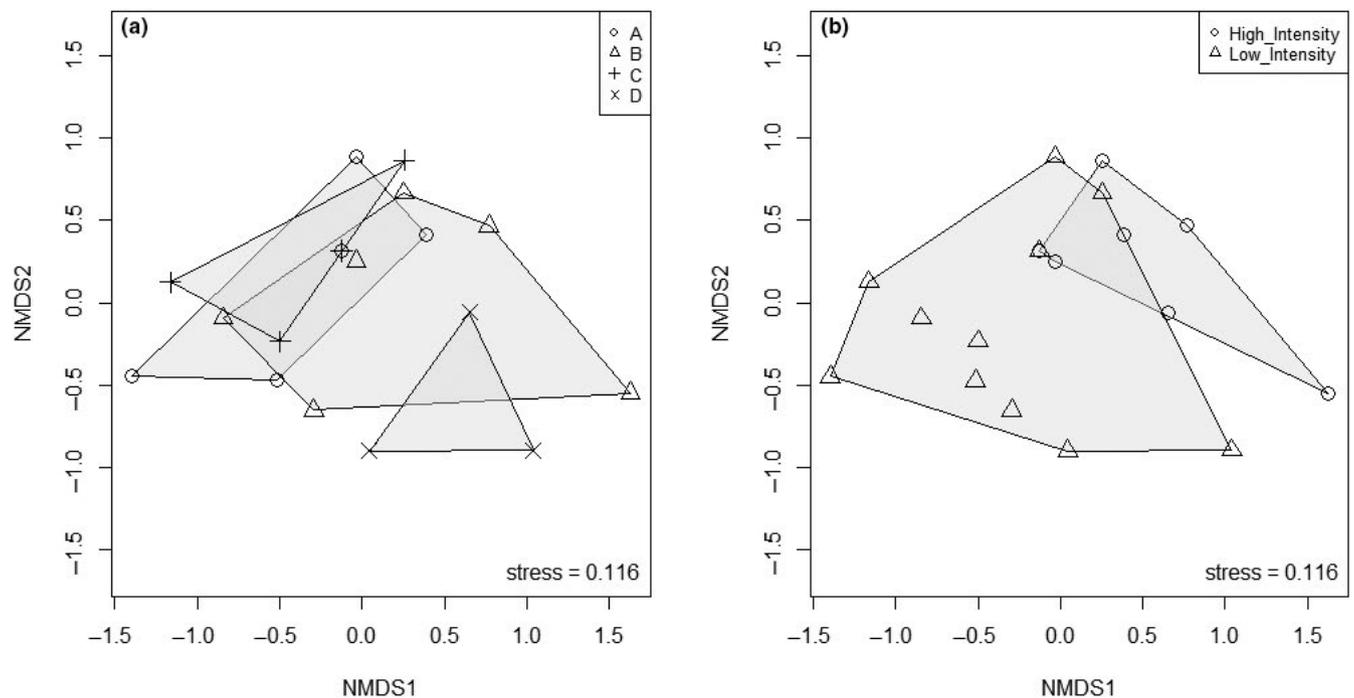
FIGURE 3 Butterfly community composition across sites representing (a) one of four conservation values, and (b) one of two landscape intensities. Data collected using transects

sampled in C sites in high-intensity landscapes (Table S6; Figure S9a,b). Hoverfly abundance and diversity were influenced by landscape intensity (explaining 14% and 19% of variation respectively) where both measures decreased when moving from low- to high-intensity landscapes. (Table S6; Figure S9c–f). Hoverfly pan trap measures were not influenced by floral richness and abundance (Table S7).

3.3 | Solitary bees

Solitary bees were found in 16 sites with numbers too low for further statistical analysis (transect: richness [4], abundance [46]; pan trap: richness [6], abundance [41]; Table S2). Eight species were recorded representing 10% of species in Ireland (77 species).

3.4 | Butterflies

Butterfly communities differed between landscape intensities ($R = 0.18$, $p = 0.05$) but not conservation value ($R = -0.022$, $p = 0.53$; Figure 3). Richness, abundance and diversity were explained by null models (Figure S10; Table S8).

4 | DISCUSSION

Conserving farmland biodiversity is increasingly important to ensure continued ecosystem service provision. SNGs are important reservoirs of plant and pollinator diversity globally (Öckinger & Smith, 2007) and are therefore key targets for biodiversity

conservation on farmland. RBAPS are a form of AES developed with the aim of conserving biodiversity and providing environmental benefits, which have received positive recognition for directly linking payments with delivery of environmental outcomes and providing benefits to target groups (O'Rourke & Finn, 2020). However, it was previously unknown how these local-scale schemes affected non-target groups like pollinators, and whether scale of management is an important consideration. We show that management at local and landscape scales can impact SNG pollinator communities with different taxa responding differently. Field-scale management through an RBAPS based on conserving SNG was demonstrated to be beneficial to non-target taxa including bumblebees, whereas low-intensity landscape-scale management benefited bumblebees, butterflies and hoverflies.

Bumblebees had positive relationships with field-scale management that promotes floral diversity and abundance. These findings complement Albrecht et al. (2007), which found that a Swiss AES benefited wild bees by promoting a higher diversity of floral resources. As different bumblebee species visit different plants for pollen and nectar resulting in niche partitioning due to tongue length (Goulson & Darvill, 2004), it is unsurprising that fields with diverse plant communities supported more diverse bumblebee communities (Ebeling et al., 2008; Potts et al., 2009). Flower abundance is important for bumblebees as more flowers throughout the flight season ensure sufficient pollen is available to produce reproductives (Bäckman & Tiainen, 2002; Lye et al., 2009). Landscape type was related to bumblebee diversity, which may suggest that diverse landscapes provide suitable nesting and hibernation sites for queens with different nesting requirements (Lye et al., 2009; Neokosmidis et al., 2018). This study was conducted when early emerging queens had established their nests; therefore, we do not have information on foraging queens or nest searching behaviour to interpret this further. Future studies looking at RBAPSs measures that would investigate queen nesting are encouraged.

Hoverflies were affected by the landscape, with higher abundance and diversity in low-intensity landscapes. Previous studies have also found hoverflies responding to wider landscape context (Power et al., 2016; Schirmel et al., 2018). Although hoverfly adults feed on pollen and nectar, larval feeding behaviour is variable (Speight, 2008). Complex, low-intensity landscapes are likely to have higher microhabitat diversity providing necessary larval resource patches (Moquet et al., 2018; Schirmel et al., 2018). Future research should assess if other landscape features, which support larval requirements, are impacted by RBAPSs to understand how different management actions affect hoverfly communities and provide relevant management recommendations. Our results indicate that sites with high floral abundances had higher hoverfly abundance. Adult female hoverflies use pollen to aid in egg maturation with many species being polylectic with unspecialised mouthparts or short proboscis to visit open flowers where pollen is easily accessible (Speight, 2008). Therefore, it could be that higher abundances of open flowers ensure enough food is available, and thus are more attractive to hoverflies.

Solitary bee datasets were too small for meaningful analysis. Using similar sampling methods in South-East Ireland, Stanley and Stout (2013) found nine solitary bee species across 10 sites of intensively managed grasslands with low floral diversity. This indicates that although the Burren is florally diverse and provides food resources to support solitary bees, there may be other factors limiting diversity. These could include lack of nesting sites in shallow rocky soils close to forage sources (Westrich, 1996) or, wetter and windier weather in the west of Ireland (Else, 2018; Walsh, 2012). These explanations are speculative with more research needed to understand why solitary bee populations appear smaller in the Burren compared to less florally diverse regions in Ireland.

Although 49% of Irish butterfly species were represented in our study, abundance and richness per site was very low, and therefore, we could not identify drivers of diversity. Many sites were open and/or exposed, and butterflies often prefer linear habitats like hedgerows that offer shelter (Dover et al., 1997). We therefore suggest future work on linear habitat management to better understand what actions would benefit butterfly diversity. Community composition displayed some difference between high- and low-intensity landscapes, but not at local scale. Butterflies can be influenced by larval host plant availability as well as forage, and these requirements differ between specialists and generalists (Krämer et al., 2012). Low-intensity landscapes could provide more suitable larval host plants and adult nectar sites, which has been demonstrated elsewhere (Cole et al., 2017; Ouin et al., 2004).

Results-based agri-environmental payment schemes are a new type of scheme promoted across Europe where farmers are paid for the environmental results they deliver (Byrne et al., 2018). Biodiversity improvement is often a key target, but for practical implementation, it needs to be easily measurable, and therefore, many RBAPSs use indicator groups. We find that plant indicators used for assessing biodiversity benefits can represent patterns in bumblebee abundance, richness and diversity, but not for other pollinator groups. This complements the findings of Tonietto et al. (2017) where habitat restoration, even if focused on plant communities, was beneficial to bumblebees. This also shows that other pollinator groups may require more focused AES measures to enhance their communities (Kohler et al., 2007). For other pollinator groups, we suggest the development of additional metrics/indicators that can be easily measured and incorporated into an RBAPS scoring system. Links between indicators and wider biodiversity is likely to be context specific (McMahon et al., 2012); therefore, we suggest similar studies across other schemes to ensure they are achieving wider biodiversity aims.

It is suggested that placing AESs in high-quality landscapes could enhance the success of field-scale measures. Although this has been demonstrated elsewhere (Kennedy et al., 2013; Rundlöf, Bengtsson, et al., 2008), we did not find any strong evidence of this. This study was carried out in a biodiversity hotspot (Parr et al., 2009; Santorum & Breen, 2005), and with a smaller number of sampling sites, we found more bumblebee and butterfly species than comparable studies elsewhere in Ireland and Scotland (Cole et al., 2017; Stanley &

Stout, 2013). Therefore, it is likely that this region is of good quality for pollinators and even high-intensity landscapes provided sufficient resources, which might explain stronger effects seen in more intensified landscapes. Nevertheless, this indicates that pre-existing SNGs could act as important pollinator population sources (Öckinger & Smith, 2007).

A potential caveat of this study is using landscape categories instead of more informative continuous variables. We designed this experiment using categories because of the constraint on the number of sites we could sample, and the categorical nature of our management variable (Text S2). While using categories is simplistic, it does give a good indication whether any effects at the edges of a gradient exist and is widely used in literature (Klein et al., 2012; Rundlöf, Bengtsson, et al., 2008; Rundlöf, Nilsson, et al., 2008; Shinohara et al., 2019). We suggest future research build upon this approach by characterising semi-natural vegetation around sites at different spatial scales to further investigate relationships between pollinator communities, RBAPS and spatial scale.

5 | CONCLUSIONS

RBAPSs are a useful tool to deliver AESs on farmland. Our results suggest that they can be useful in conserving some components of non-target biodiversity, and with some additional considerations could benefit more groups. In terms of SNG management within AESs, our results suggest that (a) measures implemented to benefit plant communities can have indirect benefits for non-target taxa like bumblebees, (b) different pollinator taxa respond differently to management, suggesting a variety of measures may capture other aspects of biodiversity and (c) local and landscape management should be considered to benefit a range of pollinator taxa. For example, including a landscape metric that assesses SNG proportion across a farm where farms with higher SNG proportions receive higher scores and/or a metric that assesses ecological connectivity between SNG patches throughout the farm (i.e. do SNGs occur adjacent to each other and are they connected to each other by other semi-natural habitats like hedgerows). The usefulness of RBAPSs for the conservation of wider biodiversity could be enhanced by including measures to represent a wider number of species. The results of this study offer insights into how AESs can increase their effectiveness for grassland pollinator communities. With future opportunities to improve AESs through EU CAP reform, our study can be used to inform AESs to promote pollinator conservation.

CONFLICT OF INTEREST
None of the authors have a conflict of interest.

AUTHORS' CONTRIBUTIONS
M.L. and D.A.S. conceived and designed the research; M.L. collected and analysed the data; M.L. and D.A.S. led the writing of the manuscript and gave final approval for publication.

DATA AVAILABILITY STATEMENT
Data available via the Dryad Digital Repository https://doi.org/10.5061/dryad.47d7wm3dd (Larkin & Stanly, 2021).

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REFERENCES
Albrecht, M., Duelli, P., Müller, C., Kleijn, D., & Schmid, B. (2007). The Swiss agri-environment scheme enhances pollinator diversity and plant reproductive success in nearby intensively managed farmland. *Journal of Applied Ecology*, *44*, 813–822. https://doi.org/10.1111/j.1365-2664.2007.01306.x

Bäckman, J.-P.-C., & Tiainen, J. (2002). Habitat quality of field margins in a Finnish farmland area for bumblebees (Hymenoptera: Bombus and Psithyrus). *Agriculture, Ecosystems & Environment*, *89*, 53–68. https://doi.org/10.1016/S0167-8809(01)00318-8

Ballare, K. M., Neff, J. L., Ruppel, R., & Jha, S. (2019). Multi-scalar drivers of biodiversity: Local management mediates wild bee community response to regional urbanization. *Ecological Applications*, *29*, e01869. https://doi.org/10.1002/eap.1869

Batary, P., Baldi, A., Kleijn, D., & Tscharntke, T. (2011). Landscape-moderated biodiversity effects of agri-environmental management: A meta-analysis. *Proceedings of the Royal Society B-Biological Sciences*, *278*, 1894–1902. https://doi.org/10.1098/rspb.2010.1923

Batáry, P., Dicks, L. V., Kleijn, D., & Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology*, *29*, 1006–1016. https://doi.org/10.1111/cobi.12536

Byrne, D., Astrain, M. C., Beaufoy, G., Berastegi, A., Bleasdale, C., David, A., Copland, A., Dunford, B., Edge, R., Finney, K., Iragui Yoldi, U., Jones, G., Lopez Rodriguez, F., Maher, C., Moran, J., McLoughlin, D., & O'Donoghue, B. (2018). *Non-technical summary: Results-based agri-environment pilot schemes in Ireland and Spain*. Report prepared for the European Union, agreement no. 07.027722/2014/697042/SUB/B2.

Carvell, C. (2002). Habitat use and conservation of bumblebees (Bombus spp.) under different grassland management regimes. *Biological Conservation*, *103*, 33–49. https://doi.org/10.1016/S0006-3207(01)00114-8

Carvell, C., Meek, W. R., Pywell, R. F., Goulson, D., & Nowakowski, M. (2007). Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins. *Journal of Applied Ecology*, *44*, 29–40. https://doi.org/10.1111/j.1365-2664.2006.01249.x

Chaplin, S., Robinson, V., LePage, A., Keep, H., Le Cocq, J., Ward, D., Hicks, D., & Scholz, E. (2019). *Pilot results based payment approaches*

for agri-environment schemes in arable and upland grassland systems in England. Final Report to the European Commission. Natural England and Yorkshire Dales National Park Authority.

Cohen, H., Philpott, S. M., Liere, H., Lin, B. B., & Jha, S. (2021). The relationship between pollinator community and pollination services is mediated by floral abundance in urban landscapes. *Urban Ecosystems*, *24*, 275–290.

Cole, L. J., Brocklehurst, S., Robertson, D., Harrison, W., & McCracken, D. I. (2017). Exploring the interactions between resource availability and the utilisation of semi-natural habitats by insect pollinators in an intensive agricultural landscape. *Agriculture, Ecosystems & Environment*, *246*, 157–167. https://doi.org/10.1016/j.agee.2017.05.007

Dicks, L. V., Corbet, S. A., & Pywell, R. F. (2002). Compartmentalization in plant-insect flower visitor webs. *Journal of Animal Ecology*, *71*, 32–43. https://www.jstor.org/stable/2693402

Dover, J. W., Sparks, T. H., & Greatorex-Davies, J. N. (1997). The importance of shelter for butterflies in open landscapes. *Journal of Insect Conservation*, *1*, 89–97.

Dunford, B. (2001). *The Impact of agricultural practices on the natural heritage of the Burren uplands, Co. Clare*. University of Ireland Dublin.

Ebeling, A., Klein, A.-M., Schumacher, J., Weisser, W. W., & Tscharntke, T. (2008). How does plant richness affect pollinator richness and temporal stability of flower visits? *Oikos*, *117*, 1808–1815.

Ekroos, J., Olsson, O., Rundlöf, M., Wätzold, F., & Smith, H. G. (2014). Optimizing agri-environment schemes for biodiversity, ecosystem services or both? *Biological Conservation*, *172*, 65–71.

Else, G. R. (2018). *Handbook of the bees of the British Isles*. Ray Society.

Feehan, J., Gillmor, D. A., & Culleton, N. (2005). Effects of an agri-environment scheme on farmland biodiversity in Ireland. *Agriculture, Ecosystems & Environment*, *107*, 275–286. https://doi.org/10.1016/j.agee.2004.10.024

Finch, T., Culleton, E., & Diamond, S. (1971). *Soils of County Clare. Bulletin No. 23*. An Foras Talúntais Dublin.

Fitzpatrick, Ú., Murray, T. E., Paxton, R. J., Breen, J., Cotton, D., Santorum, V., & Brown, M. J. F. (2007). Rarity and decline in bumblebees – A test of causes and correlates in the Irish fauna. *Biological Conservation*, *136*, 185–194. https://doi.org/10.1016/j.biocon.2006.11.012

Goulson, D., & Darvill, B. (2004). Niche overlap and diet breadth in bumblebees are rare species more specialized in their choice of flowers? *Apidologie*, *35*, 55–63.

Herzon, I., Birge, T., Allen, B., Povellato, A., Vanni, F., Hart, K., Radley, G., Tucker, G., Keenleyside, C., Oppermann, R., Underwood, E., Poux, X., Beaufoy, G., & Pražan, J. (2018). Time to look for evidence: Results-based approach to biodiversity conservation on farmland in Europe. *Land Use Policy*, *71*, 347–354. https://doi.org/10.1016/j.landusepol.2017.12.011

Hothorn, T., Bretz, F., & Westfall, P. (2008). Simultaneous inference in general parametric models. *Biometrical Journal*, *50*(3), 346–363. https://doi.org/10.1002/bimj.200810425

Hsieh, T. C., Ma, K. H., & Chao, A. (2019). *iNEXT: Interpolation and extrapolation for species diversity*. R package version 2.0.19.

IPBES. (2016). *The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/zenodo.3402856

Keenleyside, C., Radley, G., Tucker, G., Underwood, E., Hart, K., Allen, B., & Menadue, H. (2014). *Results-based payments for biodiversity guidance handbook: Designing and implementing results-based agri-environment schemes 2014–20*. Prepared for the European Commission, DG Environment, Institute for European Environmental Policy.

Kennedy, C. M., Lonsdorf, E., Neel, M. C., Williams, N. M., Ricketts, T. H., Winfree, R., Bommarco, R., Brittain, C., Burley, A. L., Cariveau, D., Carvalheiro, L. G., Chacoff, N. P., Cunningham, S. A., Danforth, B. N., Dudenhöffer, J. H., Elle, E., Gaines, H. R., Garibaldi, L. A., Gratton, C., … Kremen, C. (2013). A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecology Letters*, *16*, 584–599. https://doi.org/10.1111/ele.12082

Kleijn, D., Baquero, R. A., Clough, Y., Díaz, M., Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E. J. P., Steffen-Dewenter, I., Tscharntke, T., Verhulst, J., West, T. M., & Yela, J. L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, *9*, 243–254. https://doi.org/10.1111/j.1461-0248.2005.00869.x

Klein, A.-M., Brittain, C., Hendrix, S. D., Thorp, R., Williams, N., & Kremen, C. (2012). Wild pollination services to California almond rely on semi-natural habitat. *Journal of Applied Ecology*, *49*, 723–732. https://doi.org/10.1111/j.1365-2664.2012.02144.x

Klein, A.-M., Vaissière, B. E., Cane, J. H., Steffen-Dewenter, I., Cunningham, S. A., Kremen, C., & Tscharntke, T. (2007). Importance of pollinators in changing landscapes for world crops. *Proceedings: Biological Sciences*, *274*, 303–313.

Kohler, F., Verhulst, J., Knop, E., Herzog, F., & Kleijn, D. (2007). Indirect effects of grassland extensification schemes on pollinators in two contrasting European countries. *Biological Conservation*, *135*, 302–307. https://doi.org/10.1016/j.biocon.2006.10.037

Krämer, B., Poniatowski, D., & Fartmann, T. (2012). Effects of landscape and habitat quality on butterfly communities in pre-alpine calcareous grasslands. *Biological Conservation*, *152*, 253–261. https://doi.org/10.1016/j.biocon.2012.03.038

Larkin, M., & Stanly, D. A. (2021). Data from: Impacts of management at a local and landscape scale on pollinators in semi-natural grasslands. *Dryad Digital Repository*, https://doi.org/10.5061/dryad.47d7wm3dd

Lye, G., Park, K., Osborne, J., Holland, J., & Goulson, D. (2009). Assessing the value of rural stewardship schemes for providing foraging resources and nesting habitat for bumblebee queens (Hymenoptera: Apidae). *Biological Conservation*, *142*, 2023–2032. https://doi.org/10.1016/j.biocon.2009.03.032

McCune, B. (2002). *Analysis of ecological communities*. MjM Software Design.

McLoughlin, D. (2018). *Pilot results-based agri-environment measures in Ireland and Navarra; End of project technical synthesis report*. A report published for the European Forum on Nature Conservation and Pastoralism.

McMahon, B. J., Anderson, A., Carnus, T., Helden, A. J., Kelly-Quinn, M., Maki, A., Sheridan, H., & Purvis, G. (2012). Different bioindicators measured at different spatial scales vary in their response to agricultural intensity. *Ecological Indicators*, *18*, 676–683. https://doi.org/10.1016/j.ecolind.2012.01.013

Moquet, L., Laurent, E., Bacchetta, R., & Jacquemart, A.-L. (2018). Conservation of hoverflies (Diptera, Syrphidae) requires complementary resources at the landscape and local scales. *Insect Conservation and Diversity*, *11*, 72–87. https://doi.org/10.1111/icad.12245

Neokosmidis, L., Tscheulin, T., Devalez, J., & Petanidou, T. (2018). Landscape spatial configuration is a key driver of wild bee demographics. *Insect Science*, *25*, 172–182. https://doi.org/10.1111/1744-7917.12383

Öckinger, E., & Smith, H. G. (2007). Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes. *Journal of Applied Ecology*, *44*, 50–59. https://doi.org/10.1111/j.1365-2664.2006.01250.x

Oksanen, J., Blanchet, G., Kindt, R., Legendre, P., Minchin, P., O'Hara, R., Simpson, G., Solymos, P., Stevens, M., & Wagner, H. (2019). *Vegan: Community ecology package*. R package version 2.5-5. Retrieved from https://CRAN.R-project.org/package=vegan

Ollerton, J., Winfree, R., & Tarrant, S. (2011). How many flowering plants are pollinated by animals? *Oikos*, *120*, 321–326.

O'Rourke, E., & Finn, J. A. (2020). *Farming for nature: The role of results-based payments*. Teagasc and National Parks and Wildlife Service (NPWS).

Osborne, J. L., Martin, A. P., Carreck, N. L., Swain, J. L., Knight, M. E., Goulson, D., Hale, R. J., & Sanderson, R. A. (2008). Bumblebee flight distances in relation to the forage landscape. *Journal of Animal Ecology*, *77*, 406–415. https://doi.org/10.1111/j.1365-2656.2007.01333.x

Ouin, A., Aviron, S., Dover, J., & Burel, F. (2004). Complementation/supplementation of resources for butterflies in agricultural landscapes. *Agriculture, Ecosystems & Environment*, *103*, 473–479. https://doi.org/10.1016/j.agee.2003.11.003

Page, N., Constantinescu, M., Demeter, L., Keenleyside, C., Oppermann, R., Popa, R., & Sutcliffe, L. (2019). *Nontechnical summary: Results-based agri-environment schemes for support of broad biodiversity at landscape scale in Transylvanian high nature value farmland, Romania.* Report prepared for the European Commission, Agreement No. 07.027722/2014/697044/SUB/B2.

Parnell, J., Curtis, T., & Cullen, E. (2012). *Webbs: An Irish flora.* Cork University Press.

Parr, S., Dunford, B., & Ó Conchúir, R. (2009). *Grasslands of the Burren, Western Ireland.* Grasslands in Europe of High Nature Value, KNNV Publishing.

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: Trends, impacts and drivers. *Trends in Ecology & Evolution*, *25*, 345–353. https://doi.org/10.1016/j.tree.2010.01.007

Potts, S. G., Woodcock, B. A., Roberts, S. P. M., Tscheulin, T., Pilgrim, E. S., Brown, V. K., & Tallowin, J. R. (2009). Enhancing pollinator biodiversity in intensive grasslands. *Journal of Applied Ecology*, *46*, 369–379. https://doi.org/10.1111/j.1365-2664.2009.01609.x

Power, E. F., Jackson, Z., & Stout, J. C. (2016). Organic farming and landscape factors affect abundance and richness of hoverflies (Diptera, Syrphidae) in grasslands. *Insect Conservation and Diversity*, *9*, 244–253. https://doi.org/10.1111/icad.12163

R Core Team. (2019). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing.

Ramankutty, N., Evan, A. T., Monfreda, C., & Foley, J. A. (2008). Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochemical Cycles*, *22*. https://doi.org/10.1029/2007GB002952

Redhead, J. W., Dreier, S., Bourke, A. F. G., Heard, M. S., Jordan, W. C., Sumner, S., Wang, J., & Carvell, C. (2016). Effects of habitat composition and landscape structure on worker foraging distances of five bumble bee species. *Ecological Applications*, *26*, 726–739. https://doi.org/10.1890/15-0546

Robinson, R. A., & Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, *39*, 157–176. https://doi.org/10.1046/j.1365-2664.2002.00695.x

Roth, T., Amrhein, V., Peter, B., & Weber, D. (2008). A Swiss agri-environment scheme effectively enhances species richness for some taxa over time. *Agriculture, Ecosystems & Environment*, *125*, 167–172. https://doi.org/10.1016/j.agee.2007.12.012

Rundlöf, M., Bengtsson, J., & Smith, H. G. (2008). Local and landscape effects of organic farming on butterfly species richness and abundance. *Journal of Applied Ecology*, *45*, 813–820. https://doi.org/10.1111/j.1365-2664.2007.01448.x

Rundlöf, M., Nilsson, H., & Smith, H. G. (2008). Interacting effects of farming practice and landscape context on bumble bees. *Biological Conservation*, *141*, 417–426. https://doi.org/10.1016/j.biocon.2007.10.011

Santorum, V., & Breen, J. (2005). Bumblebee diversity on Irish farmland. *Irish Journal of Agri-Environmental Research*, *4*, 79–90.

Schirmel, J., Albrecht, M., Bauer, P.-M., Sutter, L., Pfister, S. C., & Entling, M. H. (2018). Landscape complexity promotes hoverflies across different types of semi-natural habitats in farmland. *Journal of Applied Ecology*, *55*, 1747–1758. https://doi.org/10.1111/1365-2664.13095

Shinohara, N., Uchida, K., & Yoshida, T. (2019). Contrasting effects of land-use changes on herbivory and pollination networks. *Ecology and Evolution*, *9*, 13585–13595. https://doi.org/10.1002/ece3.5814

Speight, M. C. (2008). *Database of Irish syrphidae (Diptera).* National Parks and Wildlife Service.

Stanley, D. A., & Stout, J. C. (2013). Quantifying the impacts of bioenergy crops on pollinating insect abundance and diversity: A field-scale evaluation reveals taxon-specific responses. *Journal of Applied Ecology*, *50*, 335–344. https://doi.org/10.1111/1365-2664.12060

Steffan-Dewenter, I., Münzenberg, U., Bürger, C., Thies, C., & Tscharntke, T. (2002). Scale-dependent effects of landscape context on three pollinator guilds. *Ecology*, *83*, 1421–1432. https://doi.org/10.1890/0012-9658(2002)083[1421:SDEOLC]2.0.CO;2

Stubbs, A. E., & Falk, S. J. (2002). *British hoverflies: An illustrated identification guide.* British Entomological and Natural History Society, Dinton Pastures Country Park, Davis St, Hurst, UK, Wokingham, UK.

Tonietto, R. K., Ascher, J. S., & Larkin, D. J. (2017). Bee communities along a prairie restoration chronosequence: Similar abundance and diversity, distinct composition. *Ecological Applications*, *27*, 705–717. https://doi.org/10.1002/eap.1481

Walsh, S. (2012). *A summary of climate averages for Ireland 1981–2010.* Met Éireann.

Webb, D. A., & Scannell, M. J. P. (1983). *Flora of Connemara and the Burren.* Cambridge University Press.

Westphal, C., Bommarco, R., Carré, G., Lamborn, E., Morison, N., Petanidou, T., Potts, S. G., Roberts, S. P. M., Szentgyörgyi, H., Tscheulin, T., Vaissière, B. E., Woyciechowski, M., Biesmeijer, J. C., Kunin, W. E., Settele, J., & Steffan-Dewenter, I. (2008). Measuring bee diversity in different European habitats and biogeographical regions. *Ecological Monographs*, *78*, 653–671.

Westrich, P. (1996). Habitat requirements of central European bees and the problems of partial habitats. In A. Matheson, S. L. Buchmann, C. O'Toole, P. Westrich, & I. H. Williams (Eds.), *Conservation of bees* (pp. 1–16). Linnean Society Symposium Series. Academic Press Limited.

Zurbuchen, A., Landert, L., Klaiber, J., Müller, A., Hein, S., & Dorn, S. (2010). Maximum foraging ranges in solitary bees: Only few individuals have the capability to cover long foraging distances. *Biological Conservation*, *143*, 669–676. https://doi.org/10.1016/j.biocon.2009.12.003

Zuur, A. F. (2009). *Mixed effects models and extensions in ecology with R.* Springer.

SUPPORTING INFORMATION

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