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A little does a lot: Can small-scale planting for pollinators make a difference?

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A R T I C L E I N F O

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A B S T R A C T

Insect pollinators are declining globally as a result of the anthropogenic pressures that have destroyed native habitats and eroded ecosystems. These declines have been associated with agricultural productivity losses, threatening food security. Efforts to restore habitat for pollinators are underway, emphasizing large-scale habitat creation like wildflower strips, yet ignoring the impact of smaller or more isolated patch-creation.

A meta-analysis of 31 independent published studies assessed the effect of scale of pollinator planting interventions (herbaceous strips, hedgerows, fertiliser/grazing/mowing control). We assessed pollinator species richness and abundance against size of intervention and type. Pollinator conservation interventions increased species richness and abundance in almost all of the studies examined, with the greatest increases in pollinator ecological metrics seen from hedgerows covering 40 m^2 and herbaceous interventions at 500 m^2.

We then analysed results from a 5-year study that deployed small pollinator habitats (30 m^2) at community gardens and farms (<150,000 m^2) practicing organic methods in the Pacific Northwest US. Small additions to pollinator resources had a significant local impact on pollinator abundance, but this effect was lost when these relatively small additions were introduced to sites in larger landscapes (>150,000 m^2).

Together, we show that small interventions (~500 m^2) can significantly benefit pollinators, but only when sufficiently densely distributed at a landscape level. Though we understand the effects of single interventions at various scales, future research is needed to understand how these relatively small interventions act cumulatively at a landscape scale, and within this context whether larger areas are still needed for some species. Nonetheless, these preliminary data are promising, and may play an important role in convincing smaller landowners to act to preserve insect pollinators.

1. Introduction

Pollinators are a crucial component of natural ecosystems, with an estimated 87.5% of all flowering plants globally relying on pollination (Cranmer et al., 2012). Insects are globally the most prolific pollinators (Rader et al., 2016; vanEngelsdorp and Meixner, 2010), and like many insect clades are in a state of global population and diversity decline (Montgomery et al., 2020). In the UK, historical records show that since 1980, 40 insect pollinator species have already gone extinct (Balfour et al., 2018). Bee species richness significantly declined in 52% of UK sites and 23% in the US studied in the same period (Biesmeijer et al., 2006; Koh et al., 2016). In response to widespread public concern about the future of these ecologically and economically important animals, in 2015 the US Government released a national strategy to support pollinators and in 2018 the European Union launched the EU Pollinators Initiative (Bloom et al., 2022a; EU, 2018).

Pollinator declines are the result of a suite of complex interactions (Goulson et al., 2015), including a lack of food sources, spreading diseases, climate change, increasing pesticide use and the spread of invasive species (Donkersley et al., 2020; Rader et al., 2016;). Habitat loss (explicitly agricultural intensification) sits at the centre of this nexus of factors as a major driver of pollinator decline (Senapathi et al., 2017). In England and Wales, key wild pollinator habitats such as semi-natural grasslands and heathlands have seen a 97% decline since the 1930 s and now makes up only 4% of the total grassland area (Fuller, 1987; Gimingham, 1994). Conventional understanding is that significant

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investment in conservation and landscape restoration is needed to address and possibly reverse pollinator declines (Brown et al., 2016), but only in combination with reduction in the drivers of these declines: agriculture (Langlois et al., 2020; Morrison et al., 2020).

Conventional pollinator conservation mainly focuses on large, national-scale strategies, such as the EU-funded agri-environment schemes under Common Agricultural Policy. These encourage lower-intensity farming and recently regenerative agriculture approaches, setting aside land to introduce heterogeneity (e.g. as floral strips or hedgerows at field margins) (Baur and Schläpfer, 2018). Similar schemes also exist for the United States, New Zealand and Australia (Albrecht et al., 2020). These can achieve regenerative agriculture approaches with minimal losses in farm profit (Hanley et al., 2011, 2015). The UK also has several voluntary initiatives to combat pollinator decline, such as 'Living Landscapes' and 'B-Lines' that aim to increase the area of suitable habitat available for pollinators and provide greater landscape connectivity (Stubbs and Shardlow, 2012; Warnock and Griffiths, 2015).

Creation of large areas of conservation habitat is challenging on a practical and political level (Marshman et al., 2019; New, 2018; Schmeller et al., 2008). A great deal of effort has been put into "rewilding" strategies on abandoned agricultural land (Lasanta et al., 2015; Loth and Newton, 2018), maximising habitat quality and conservation efficiency in "edgelands" like roadside verges (Phillips et al., 2020). Focus has been on efforts like introducing wildflower strips on agricultural land (Albrecht et al., 2020; Baur and Schläpfer, 2018) or improving existing land reserved for conservation (Senapathi et al., 2017). Factors that maximise efficiency of actions and scale include: site shape, such as using corridors of land, or so-called stepping-stones that improve landscape connectivity for pollinator dispersal (Menz et al., 2011). Site conditions are also key: floral community characteristics, soil characteristics, adjacent land use and physical properties of the site and surroundings (Hopwood et al., 2015), edge effects (Olson and Andow, 2008) and disturbances such as mowing (Hobbs and Huenneke, 1992; Wojcik and Buchmann, 2012) all play an important role in habitat suitability. Site specificity plays an important role in terms of "type" of intervention: wildflower patches, sandy banks, hedgerows and mowing/grazing regime adaptations are all context-dependent and have different impacts on insect communities (Albrecht et al., 2020; Bloom, 2019; Haaland et al., 2011; Halada et al., 2011; Mader et al., 2010).

Many landowners are deterred from setting aside land: loss of yield for arable farmers (Bowman and Zilberman, 2013; Mills et al., 2018), urban home-owners having limited space (Saint Consulting, 2009), or public pressure for councils to regularly mow green spaces for sports and leisure (Turo and Gardiner, 2019) are all well-known factors. Low economic value land tends to be used for pollinator conservation with limited success seen in "no-mow" roadside verge schemes (Phillips et al., 2020). With the growing need to provide more stable habitats for pollinators with scarcer available space (Brown and Paxton, 2009; Decourtye et al., 2019; Khanna et al., 2018; Samways et al., 2020), small patches of land, urban community gardens (Makinson et al., 2017) and small private gardens (Braschler et al., 2020) may be an important resource for pollinators. The single large or several small (SLOSS) debate becomes highly relevant within the context of supporting insects through wildlife corridors (Wagner et al., 2019). Flying insects move easily between smaller patches in search of food or nesting sites (MacDonald et al., 2018). Consequently, we need to ask: can a heterogeneous anthropogenic landscape comprising small patches of conservation habitat have a significant cumulative impact on pollinator diversity (Maurer et al., 2020)? To our knowledge, very few studies have explicitly considered quantifying the benefits of smaller interventions for pollinators (i.e. <500 m^2).

In this study, we consider the spatial "tipping point" for pollinator conservation schemes, in terms of abundance and diversity gains relative to the size of intervention. First, we undertook a meta-analysis of spatially explicit pollinator conservation studies, examining whether small interventions (<500 m^2) can achieve anything more than a marginal benefit to pollinators in comparison with studies using large interventions (>500 m^2).

Next, we examine data collected from small habitat interventions (30 m^2) on community gardens and farms ranging from 5000 m^2 - 150,000 m^2. We aim to provide important contextual information on whether the size of the farm in which these "small interventions" take place in matters. Combined, these studies aim to address the question: is there any benefit to pollinators from small habitat restoration efforts, and does this effect dissipate over larger landscapes?

2. Materials and methods

2.1. Meta-analysis of "small" plots in a "big" landscape

To determine the impact of various pollinator conservation interventions, relative to the size of said intervention, we first perform a meta-analysis of existing studies in accordance with the PRISMA checklist for methods (Page et al., 2021). PRISMA item numbers are indicated where appropriate throughout the methods. We conducted a literature search of pollinator conservation studies using the Conservation Evidence database (www.conservationevidence.com) [PRIMSA 6].

The literature search was concluded by December 2020. Studies were first selected using the search term "pollinators", generating 29 action categories, totalling 581 studies (see Supplementary Materials) [PRIMSA 7]. From these, we selected only action categories that contained actions representing floral resource manipulations targeted at pollinators, reducing the number of studies to 421 (see Supplementary Materials). Duplicate studies that are covered across multiple action categories were removed, reducing the total to 330 studies. These studies were then manually screened through abstract, methods and results respectively, by the authors according to a set of seven criteria, whereby the study must:

- be conducted within the last 40 years (n = 327);
- be published in an accessible peer-reviewed format (n = 287);
- be conducted in the global north (n = 272);
- report pollinator specific results (n = 90);
- report species richness or abundance or both and plot areas used, with a control plot and at least one plot that had a conservation intervention (n = 31);
- does not take an a-priori assumption of positive response (n = 31; Section 3.1.).

To establish factors that influence the effectiveness of interventions in these studies, data were grouped: intervention types were grouped according to standard agri-environment groupings into '*herbaceous*' where different seed mixes were used; '*mowing*' involving changes to vegetation management regime; '*hedges*' where hedges were adjacent to intervention plots; '*grazing*' in which changes were made to livestock grazing patterns and '*fertilizer*' where use of fertiliser was assessed.

Following manual independent screening by two of the authors of each study in the database search results [PRIMSA 8], data for both species richness and abundance were extracted from data attached to the publications, specifying "intervention" and "control" data [PRIMSA 9]. Intervention size, site location, land use type, landscape setting (arable/pastoral and rural/urban), the state of interventions vs. controls and study specific notes were all extracted and compiled across the studies [PRIMSA 10]. The authors considered, but found no risk of bias in our assessment [PRIMSA 11/14].

Data on pollinator species richness and abundance observed on sites were extracted for this meta-analysis [PRIMSA 12]. Studies used different methods to calculate pollinator species richness or abundances such as pan-traps, in-situ observations or suction sampling. Methods that produced extreme anomalous values were discounted. In papers that recorded data over several consecutive years or took repeated samples, means were calculated for each plot and intervention [PRIMSA 13].

Given the wide range of plot sizes for most conservation interventions, to aid in interpretation the study scale was coarsely divided *post-hoc* by study plot areas into two discrete categories, where small plots were conceptually defined as < 500 m^2 and large plots ≥ 500 m^2. The boundary for the discrete categories was chosen *post-hoc* as it splits the data into two groups each with a similar number of data points. In addition to categorising by intervention type and study scale, other potential groupings were explored (e.g. land use type, rural setting vs. urban setting and arable vs. pastoral land). However, for most groupings there were too few data points in at least one category, and these groupings were therefore not included in subsequent analyses.

2.1.1. Data transformations and analysis

After these data were categorised, the raw data were transformed for comparison, to account for differences in pollinator sampling between studies [PRIMSA 20]. Several studies only focused on species from a single genus, whereas other studies included pollinators from a range of different taxonomic orders. We selected studies that used a paired intervention/control design. Changes in both species richness and abundance for each intervention were calculated as the change relative to the controls for each data point:

$$\text{Relative change} = (X_{CT} - X_{IN}) / X_{CT}$$

where X_{CT} is the control value and X_{IN} is the corresponding value in a given intervention. To calculate the saturation effects of larger plots on species richness and abundance, the relative change values were also analysed following further division by the plot size to give a relative change per m^2. Relative change values in (1) pollinator abundance and (2) species richness were analysed separately.

2.2. Experimental study of "small" plots in a "small" landscape

Next, to contextualise the results of our meta-analysis, we collected data on our own small interventions on community gardens and small diversified farms (hereafter: sites) practicing organic methods representing a range of site sizes (5000 m^2 - 150,000 m^2). Habitat interventions were randomly assigned among these sites in a paired design. Sites were paired by proximity and production practices (to control local and landscape context), with eleven sites serving as controls and receiving no treatment. The remaining 11 sites received a habitat intervention, consisting of a floral strip, bare ground, and cavity nest. This habitat was installed in the margin of the site, within five meters of cultivated crops. These sites are found in the Pacific Northwest, USA (Supplementary Materials A) and have previously been described in detail by Bloom et al. (2022b).

Installation of floral strips began in June 2016. The site margin was rotary ploughed, tilled, and solarized with 6 mm greenhouse plastic. Solarization was used to reduce pressure from weeds post-planting (Jordan et al., 2018). After solarization, a mix of native perennial and annual forbs was broadcast seeded in October and November (*see* Supplementary Materials for species composition of the floral strip; Heritage Seedlings Inc., Salem, OR, USA). Floral strips were maintained by weeding, which was performed by-hand two times per year in 2017 and 2018. The bare ground habitat intervention was created to attract ground nesting bees (Mader et al., 2010) through rotary plough tillage, and then rotary ploughed again to mound the soil and create a "bee bed". This bee bed was then solarized until October/November when the floral strip was seeded and thereafter remained bare using hand weeding at the frequency described above for floral strip maintenance. The final component of the habitat intervention was a cavity-nest, used to promote stem and twig nesting bees (Bloom et al., 2018). Cavity-nests were constructed from wooden posts (hereafter: nest blocks) placed inside of shelters made from common materials found at the hardware store (e.g. plywood, screws, corrugated roofing) (*see* Bloom et al., 2018 for shelter images). Nest blocks included 8 cavity sizes (4–11 mm in diameter) at 3 depths (90–140 mm). Nests were constructed following instructions provided by the United States Department of Agriculture (USDA, 2016). Shelters were placed facing southeast to maximize exposure to the sun. Structures were installed at the time floral strips were seeded (October/November). Altogether, the dimensions of the habitat interventions were approximately 15×2 m (30 m^2), with the floral strip and bee bed measuring 10×2 and 4×2 m, respectively. Thus, a 1×2 m area within the intervention was used for the cavity-nesting structure, the ground below which was also kept bare. Site specific effects are controlled for with random effects in linear mixed effects models (see 2.2.2.).

2.2.1. Insect sampling

Bee abundance was measured three years before (2014–2016) and two years after (2017–2018) the installation of habitat. Sampling for bees was conducted three times (spring, summer, fall) at each of the 22 farm sites (*see* data DOI for exact sampling dates). Three blue vane traps (SpringStar LLC, Woodinville, WA, USA) and 15 bee bowls (5 blue, 5 yellow, and 5 white) were placed at each site to sample the bee community in each time period. Traps were placed along a 50 m transect beginning 5 m from the field margin or habitat intervention (Droege, 2008) from 07:00–17:00 at temperatures above 12°C with minimal cloud cover and wind. Bees were also netted in two 15 min bouts; one bout between 09:00 and 11:00, and another bout between 14:00 and 16:00, both beginning 5 m from the field margin/habitat installation along a serpentine transect. Thus, bees were sampled adjacent too but not from the margin or habitat intervention thereby evaluating bee abundance within the site production area. The method for serpentine transects are described in Bloom et al. (2019). For the analysis described herein, bees were identified to bumble bees, honeybees and other bees. Due to potential noise induced in data by honeybee visitation, species richness was not analysed. A prior analysis of bee species identity was conducted by Bloom (2019).

2.2.2. Data analysis

First, all data were summed across sample years and compared between sampling method (blue vane traps, bee bowls, and netting). These data were then sub-analysed summed within each sample year to identify any outlying sampling methods potentially skewing results. Bee abundance data were analysed summed across all sample years and sampling methods. Insect counts were analysed in generalised linear mixed effects models. Control variables included years since transition to organic farming, site type (community garden or diversified farm), sample year (in the relevant sub-analysis). Explanatory variables were sampling month, site scale (perimeter and area), non-linear factors in mixed effects models identified "tipping points" for where our interventions stopped being effective. Bee abundances were each identified in total, followed by each bee category individually. Results presented are those of the most parsimonious model. All analyses were performed in R 4.0.5 (R Development Core Team, 2020). Mixed effects models were carried out using the "lme4″ package (Bates et al., 2015), model results were extrapolated and plotted using the "npreg" package (Helwig, 2020).

2.3. Ethical approval and consent

Ethics approval was not required for this study.

3. Results

3.1. Meta-analysis of spatially explicit pollinator conservation studies (Europe)

The final dataset of studies included: a total of 31 independent studies (see Supplementary Materials) that reported insect pollinator biodiversity impacts from various intervention types (i.e. wildflower strips, hedgerows, no-mow schemes, etc.) that also specifically reported the size of these interventions. Of these, 18 provided data for species richness ($n = 34$) and 27 provided data for pollinator abundances ($n = 45$) [PRIMSA 16].

3.1.1. *Intervention size (quantitative)*

Large interventions (>500 m^2) resulted in significantly greater increases in species richness relative to controls ($t = -2.493$, $df = 15.191$, $p = 0.025$). The median value for small interventions (<500 m^2) was a 0.15-fold increase and not significantly different between control and treatments ($t = -0.452$, $df = 45.752$, $p = 0.653$) whereas in large plots it was much greater at a 1.2-fold increase in species richness (Fig. 1a). Increases in pollinator abundance showed a similar, though notably less extreme difference, as small interventions produced an increase of 1.4-fold relative to controls ($t = -1.436$, $df = 54.594$, $p = 0.157$) and large interventions generated a 2.1-fold increase in abundance ($t = -2.290$, $df = 29.87$, $p = 0.029$, Fig. 1b).

3.1.2. *Intervention size (qualitative)*

Compared with control plots within each study included in the meta-analysis, interventions increased pollinator species richness by 88% (Fig. 2a) and pollinator abundance by 91% of cases (Fig. 2b). Qualitatively: for species richness, the eight plots in which interventions had the greatest effect on pollinators were all greater than or equal to 500 m^2 in area (Fig. 2a). Similarly, for pollinator abundance eight out of the 11 sites in which interventions had a substantial effect were larger than or equal to 500 m^2 (Fig. 2b).

Although species richness and abundance declined in response to interventions in five studies, the decreases in both species richness and abundance were never more than 0.37-fold (Fig. 2b). Qualitatively: the greatest increase in species richness, at 12.7-fold was at a 500 m^2 herbaceous intervention, beyond this tipping point species richness gains are less by scale. By contrast, the greatest increase in species richness for all other intervention types was 1.73-fold. Similarly, the largest increases in pollinator abundance occurred in response to herbaceous interventions with increases of up to 7.4-fold, whereas all other interventions only achieved a four-fold maximum increase.

3.1.3. *Tipping points*

At interventions larger than 500 m^2, there was little evidence that further increases in scale produced additional scalar benefits to pollinator abundance or species richness. At plot sizes above 4220 m^2 increases to species richness (Fig. 2a) and abundance (Fig. 2b) in response to conservation interventions ceased.

Interventions did not necessarily generate strong increases in pollinator species richness or abundance at all large plots, as both species richness and abundance declined in response to interventions in one plot larger than 500 m^2 (Fig. 2). The largest observed increase in pollinator

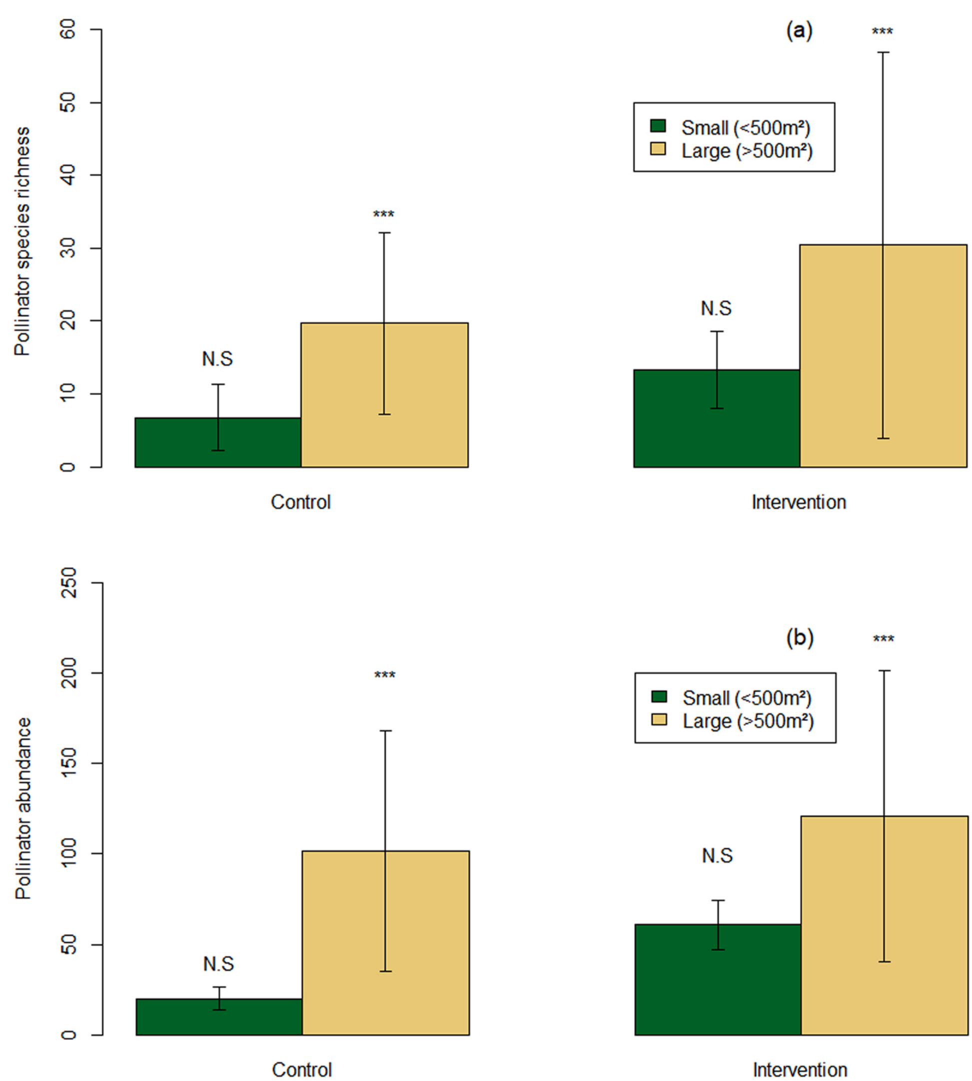

Fig. 1. Box plots assessing the median and range of changes in insect pollinator (a) species richness and (b) abundance of intervention plots compared to control plots for small sites (<500 m^2) and large sites (≥500 m^2).

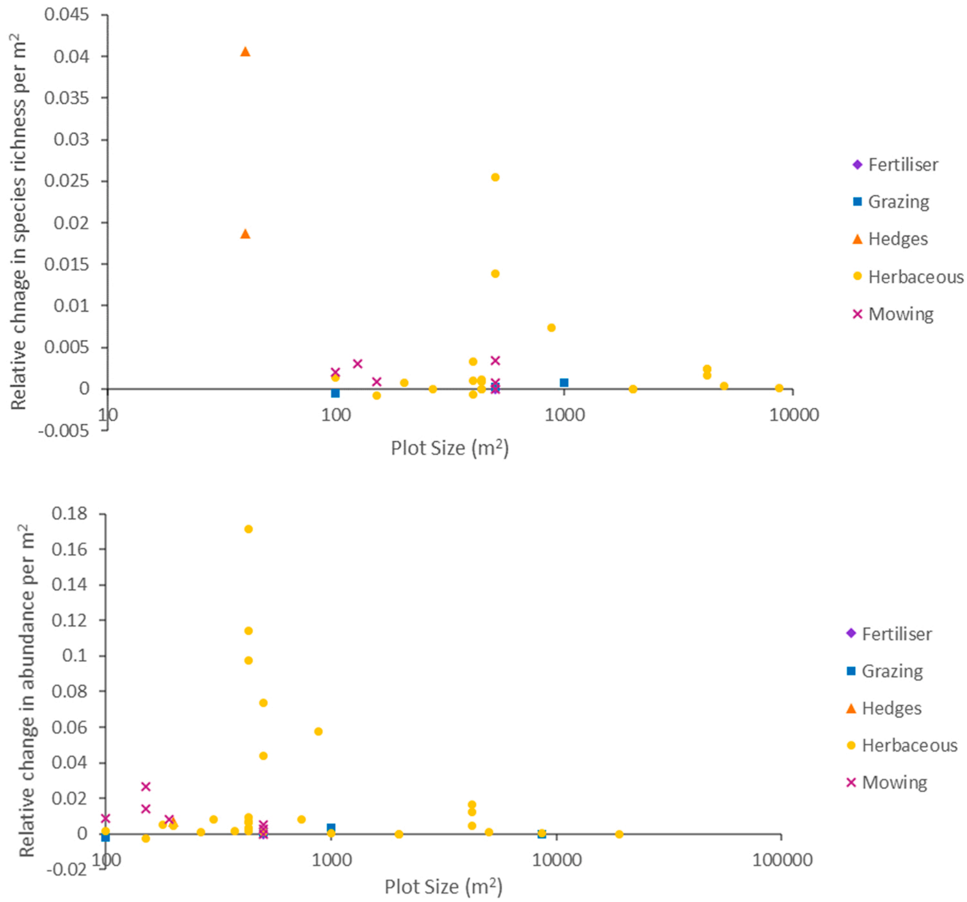

Fig. 2. Scatterplot displaying change in insect pollinator abundance for different types of intervention plots compared to control plots across different plot sizes.

abundance in one of the studies deploying plots measuring 432 m^2 (Fig. 2), which suggests an optimal "tipping point" for successful recruitment of pollinators slightly below the threshold of 500 m^2 (Fig. 2).

Linear regression of the meta-analysis data, divided into the aforementioned "large" and "small" datasets, demonstrated that species richness was significantly correlated with site size in both large ($F = 5.329$, $df = 2, 17$, $p = 0.016$) and small ($F = 7.972$, $df = 2, 45$, $p = 0.001$) groupings (Fig. 3a). Pollinator abundance was not significantly correlated with site size for the large grouping ($F = 2.386$, $df = 2, 29$, $p = 0.109$), but it was significantly correlated in the small group ($F = 3.791$, $df = 2, 55$, $p = 0.029$; Fig. 3b).

3.1.4. Intervention type relative to plot size

After transforming data to compare interventions types relative to the size of their plot areas, it was apparent that species richness per unit area values were considerably greater at 40 m^2 and 500 m^2 and abundance per unit area values were greatest from 432 to 875 m^2 (Supplementary Materials B). Per unit area, herbaceous interventions provided the greatest increases in pollinator species richness and abundance at larger plot scales (>500 m^2). However, small interventions of hedges (~40 m^2) resulted in the greatest increase in species richness (0.041-fold per m^2; Fig. 2a). Though sample sizes are too small and therefore preclude any quantitative analysis, it is noteworthy from a more qualitative analysis that mowing produced the greatest increases in pollinator

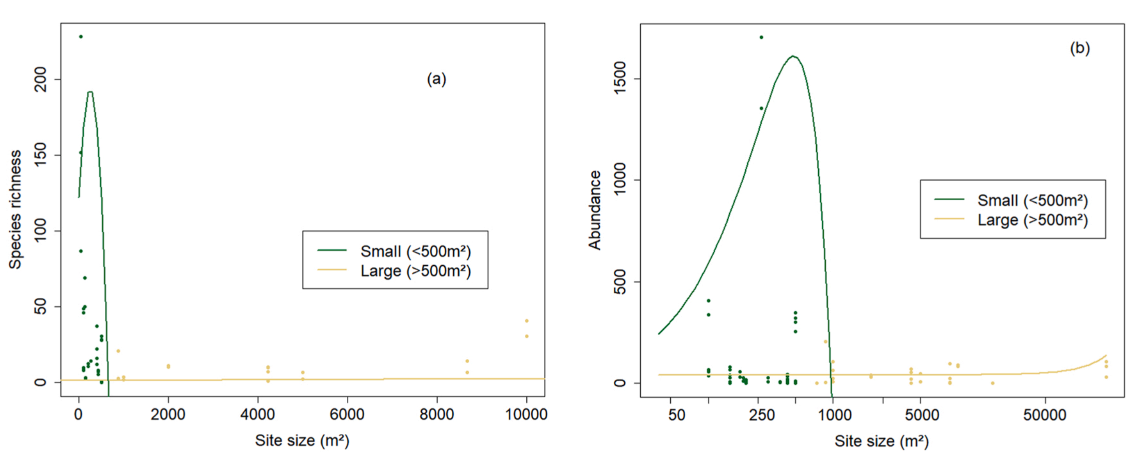

Fig. 3. Scatterplot displaying change in insect pollinator (a) species richness and (b) abundance with fitted splines of non-linear regressions against site size.

abundance on plot sizes < 432 m^2, although the increase was still more than six times smaller than in herbaceous interventions on large plots.

3.2. Habitat interventions on diversified farms and community gardens (Pacific Northwest, USA)

3.2.1. Sampling data summary

Across the five years, bumblebees were the least abundant caught using any sampling method (n = 2794), followed by honeybees (n = 3252), with the most abundant being all other bees (n = 4713). Although pollinator catch was impacted by trap method (Table 1), this effect was consistent across sample years. The mean total number of pollinators trapped across all farms in all years was 490 ± 158 (Table S1).

3.2.2. Modelling impact of interventions on pollinator abundance

The total number of bees trapped in sites that received the floral strips was greater than the control sites (n = 1099), though the increase in number of honeybees (n = 32) and bumblebees (n = 22) was lower than the increase in other bees (n = 1045).

Linear mixed effects models were constructed from pollinator sample data. Follow REML testing of models, the random effects included in the most parsimonious model were sampling method, sample year, and site type (community garden/diversified farm) nested within organic status.

Over the course of the 5 years in which the wildflower strips were managed, the abundance of pollinating insects caught in each of the farms, in each of the trap technologies deployed, increased from a total of 1360 individuals to 3550, nearly doubling during this time (Table 1).

Total pollinator abundance within the data collected showed a non-linear cubic relationship with site area (Fig. 4). This held true for both honeybee and other bee pollinator clades, but not for bumblebees (Table 2, Fig. S1), though the decreased catch rate for bumblebees may be limiting this effect (Table 1). The mixed effects model demonstrates, when habitat interventions of a fixed small size, these are effective in increasing pollinator abundance, but only when they are used in a relatively small site. When the farms start to get larger, (approaching 150,000 m^2) the effect diminishes rapidly (Fig. 4).

4. Discussion

In the study we present here, we combine two datasets on pollinator conservation and diversity in agricultural landscapes: (1) a meta-analysis of pollinator conservation studies and (2) a 5-year experiment studying the effectiveness of habitat interventions within community gardens and diversified farms to test when “small” interventions would be most effective, and whether even small farmers can contribute to promoting pollinators.

The meta-analysis of pollinator intervention studies (performed in larger agricultural landscapes) showed that small interventions (i.e. wildflower strips, hedgerows, mowing regimes, Fig. 2), above a 500 m^2 threshold value, had higher median species richness and abundance than plots below 500 m^2. However, at interventions larger than 500 m^2, there was little evidence that further increases in scale produced additional intervention benefits to pollinator abundance or species richness. Scale is an important constraint for insect pollinator diversity, and going “bigger” displays clear diminishing returns, at least directly in-situ.

When considering the other end of the “scale spectrum”, results from the experiment conducted on community gardens found that even small (~30 m^2) wildflower strips had detectable benefits to insect pollinator abundance. Although direct comparison between the two studies would be disingenuous, the evidence suggests small interventions in bigger landscapes (>50,000 m^2) had a minimal impact on pollinators. Together, these results represent important preliminary findings in the case for small scale pollinator conservation: large interventions do not necessarily mean more pollinators by scale, that small interventions are not beneficial on large farms.

The meta-analysis of “small intervention” types suggested herbaceous interventions were the most successful intervention type, in terms of pollinator diversity increases per unit area. Certainly, the conventional understanding of herbaceous interventions is that their use of more florally diverse seed mixes attracts a greater range of pollinators (Haaland et al., 2011). Both generalists and specialists benefit from more diverse mixes, many species of insect pollinators are known to prefer to feed on a range of different flowering plant species to give them a varied diet (Nichols et al., 2019). However, it is important to highlight the link between reductions in intervention size and the plant species carrying capacity: with effects like re-seeding and interspecific competition effectively countering work by land managers to introduce more wildflower diversity in these small areas (Briscoe Runquist et al., 2016; Sandau et al., 2019).

(Re)-introducing hedgerows was also an effective intervention for increasing species richness per unit area on small plots. This emphasises the importance of large, woody plant structures for pollinators (Donkersley, 2019). Hedges provide a 3-dimensional network of twigs and branches that can be ideal nesting habitat for insect pollinators, including some solitary bee species, as well as flowering hedges offering a food source for pollinators (Morandin and Kremen, 2013). Historically, hedgerows are particularly important in the UK, having an estimated 190,000 km of intact ancient hedgerows (Croxton et al., 2004), offering an example for their particular management requirements. Unlike naturally senescing wildflowers, post-harvest flailing is common practice today hedgerows are annually cut, typically between November and February (Bright and MacPherson, 2002; Britt et al., 2000). These management programs are energy and time intensive, part of an extensive cultural history with a huge array of options and techniques (Höpfl et al., 2021). Although appropriately managed hedgerows offer a substantial floral and nesting habitat resource for wild pollinators (Byrne and deldelBarco-Trillo, 2019), flailing undertaken in the interest of tidiness results in hedges becoming very reduced, and sometimes shorter than the crops they surround (Croxton et al., 2004). Wildlife-friendly hedgerows require very specific management (Höpfl et al., 2021), adopting these practices and disseminating knowledge of them has thus far limited their potential as nature-based solutions for pollinator conservation (Collier, 2021).

The results of the meta-analysis also suggested that alterations to the mowing regime were able to produce relatively large increases in pollinator abundance per unit area for small plot sizes. Changes to more pollinator-friendly mowing regimes include timing mowing after flowers had finished blooming; reducing mowing frequency, reducing the cut height (Chaudron et al., 2020) or leaving some strips uncut (Buri

Table 1
Pollinator trapping data summary, split across major identified pollinator clades, trapping methodology and sampling year.

Bumblebees				Honeybees				Other bees				All bees	
Year	Blue vane	Netting	Pan trap	Year	Blue vane	Netting	Pan trap	Year	Blue vane	Netting	Pan trap	Year	Sum
2014	93	246	10	2014	107	147	78	2014	208	217	254	2014	1360
2015	339	316	83	2015	108	584	96	2015	273	38	126	2015	1963
2016	153	297	25	2016	111	789	85	2016	165	41	107	2016	1773
2017	163	249	35	2017	88	349	73	2017	318	191	422	2017	1888
2018	239	439	92	2018	91	453	81	2018	744	348	1063	2018	3550

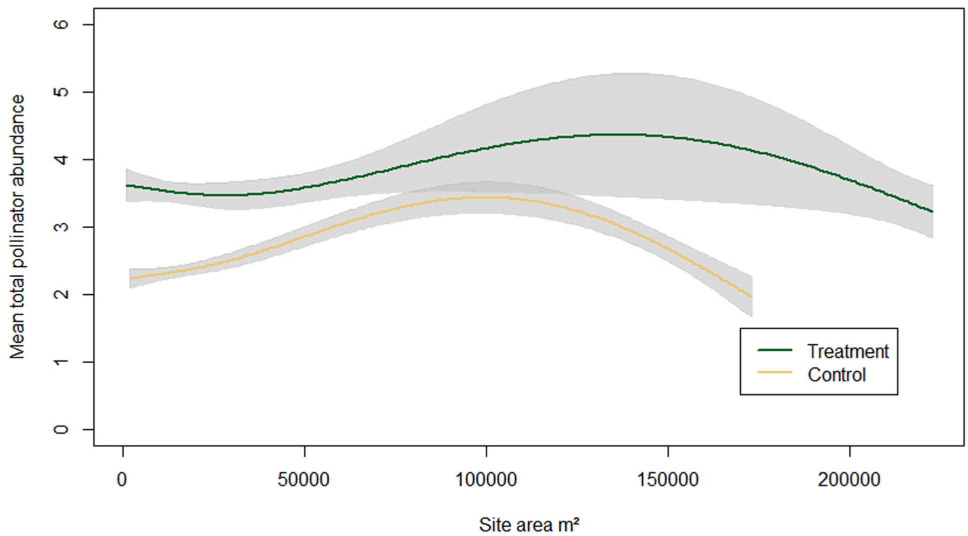

Fig. 4. Mean observed total pollinator count varies significantly with site area; smooth spline plotted based on extrapolated data from linear mixed effects model (lmer).

Table 2
Mixed effects model (lmer) statistical results on the effect of increasing farm area size on pollinator abundance.

	Total bees		Bumblebees		Honey bees		Other bees	
	b ± SE	t (P)	*b* ± SE	t (P)	*b* ± SE	t (P)	*b* ± SE	t (P)
Area	0.663 ± 0.828	0.802	1.698 ± 0.583	2.913 * *	0.317 ± 0.615	0.517	-0.065 ± 0.829	-0.079
Area^2	-2.585 ± 0.806	-3.209 * *	0.010 ± 0.568	0.018	-0.638 ± 0.598	-1.068	-2.833 ± 0.807	-3.511 * **
Area^3	2.061 ± 1.005	2.049 *	1.534 ± 0.671	2.286 *	-1.047 ± 0.752	-1.392	3.135 ± 1.021	3.071 * *
Month	-0.025 ± 0.008	-3.370 * **	-0.0681 ± 0.006	-12.081 * **	0.084 ± 0.006	15.110 * **	-0.053 ± 0.007	-7.173 * **
Intercept	1.640 ± 0.42	3.905 * **	1.078 ± 0.278	3.872 * **	-0.001 ± 0.349	-0.004	1.106 ± 0.219	5.058 * **
Observations	3187		3187		3187		3187	
Log Likelihood	-3187.079		-2249.692		-2201.955		-3124.339	
AIC	6394.158		4519.384		4423.911		6268.678	
BIC	6454.826		4580.052		4484.579		6329.347	
Note:	$^*p < 0.05$; $^{**}p < 0.01$; $^{***}p < 0.001$							

et al., 2014). This suggests that changes to mowing can limit the disruption caused to pollinators from the loss of floral resources and can even improve habitat heterogeneity resulting in increases in pollinator population sizes (Johansen et al., 2019; Lerman et al., 2018). Mowing regime alterations have the appearance of simplicity, though impediments to their widespread implementation include a widespread cultural aversion to "messy" spaces (Hoyle et al., 2017).

Based on the "tipping point" of size of intervention to biodiversity impact, areas of approximately 500 m^2 are most valuable for promoting pollinator species richness and abundance. The relatively small size of 20–500 m^2 intervention plots also means it would be easy to find available land that could be used for small scale pollinator conservation. Offset schemes like these could be used within development projects that match creation of new, high quality habitat to offset the damages associated with such development (Sonter et al., 2020). Environmental impact assessments (EIA) and life cycle impact assessments (LCIA) are commonplace worldwide, though acknowledgement of insects and insect pollinators is lacking in both EIA and LCIA frameworks that have practical implementation. Recent proposals for the inclusion of pollinators in EIA and LCIA still place emphasis on the food and farming sector, rather than urban and construction sectors (Crenna et al., 2017; Franklin and Raine, 2019; Othoniel et al., 2019). Using the pollinator conservation plots as part of offset schemes can also be employed in long-term agreements between food growers and retailers where pollinator conservation commitments are required as part of the contract (IEEP and IUCN, 2018).

Boosting pollinator diversity using flora-rich patches and planting (or even restoring) small patches of hedgerows can directly benefit landowners by creating an ecosystem with a high richness of other species (Bright and MacPherson, 2002; Britt et al., 2000; Collier, 2021). Increased bird species richness through habitat restoration programmes improves the aesthetics of properties which can increase the house value (Farmer et al., 2013). Small scale interventions are potentially more appealing to small landowners with conservation hedgerow planting costing around £ 664 per 100 m and a wildflower seed mix to cover a 500 m^2 patch costs approximately £ 71 (Staley et al., 2015). Engaging local communities to make them more aware of sustainable practices to protect pollinator diversity could also potentially lead to future citizen science projects to conduct further monitoring and research to enhance insect pollinator conservation (Saunders et al., 2018).

Our data demonstrated how variable size of intervention impacts pollinators at a landscape level. Correspondingly, we then performed an experiment using a fixed size of small intervention (wildflower strips) introduced to community gardens of varying size in the Pacific Northwest (USA) to more directly interrogate our observations from the meta-analysis, determining whether the scalar "tipping point" shifts depending on the site size it is located within. The results of this study primarily demonstrated that small interventions have demonstrable benefits to pollinator diversity on these community gardens, but only when those gardens/farms are smaller than 50,000 m^2. One conclusion we draw from this finding is that although small interventions are attractive to land managers (see below), a minimum density of them may be required to effect these larger landscapes. Previous research into the "island biogeography" effect on highly mobile insects pollinators however, highlights the importance of considering factors other than species diversity. More fragmented sections of natural habitat in agricultural landscapes can negatively impact the fitness and behaviour of bumblebees foraging in those landscapes (Maurer et al., 2020). As such, although small sections of conservation habitat can be beneficial, it is essential to consider optimal spatial configuration of these sections to avoid "doing more harm than good".

In comparison with the density of research on pollinator conservation schemes for rural (primarily agricultural) environments (Albrecht et al., 2020; Holzschuh et al., 2016), urban pollinator ecology have remained largely under-researched (Harrison et al., 2019; Hicks et al., 2016). Small-scale conservation efforts like those we have detailed here are likely more suitable for urban (or anthropogenic) environments, due to a lack of expansive areas for conservation schemes commonly found in rural space. Urban landscapes support fewer rare bee species than natural landscapes, and in many cases worldwide, urbanization and nutritional stress have collectively contributed to declines (Harrison et al., 2019; Penick et al., 2016). With significant "grass roots" efforts within communities, urban environments can support native solitary bee communities (Prendergast, 2021). High quality, diverse forage available in nearby urban environments have the potential to support rural bee species if the conditions are suitable (Garbuzov et al., 2015; Lowenstein et al., 2015).

These findings are relevant in particular to small-medium enterprises (SMEs) based in urban environments, as many of these businesses would be able to afford to plant wildflowers on 30–500 m^2 plots of land (Pimenova and Van der Vorst, 2004). Many SMEs will already own small areas of land such as the borders surrounding the company buildings or their carparks, or disused land with low economic value such as contaminated land or land without planning permission and these could be transformed into pollinator conservation plots (Kattwinkel et al., 2011). Implementing pollinator conservation interventions on SME land could also indirectly benefit the company by improving the business image and could attract investor interest (Jain et al., 2017).

Many smaller landowners are deterred from setting land aside for pollinator conservation (Mills et al., 2018; Turo and Gardiner, 2019), due to economic constraints and the perceived necessity of scale required (Fahrig, 2020). These perceptions and motivations guiding environmentalism at the small, local scale may be hindering efforts to restore and protect ecosystems (Kollmuss and Agyeman, 2002; Siegel et al., 2018). Evidence that shows "little efforts", like the small wildflower patches used in this study, may form an important part of the narrative in guiding uptake of pro-nature conservation behaviours (Barbett et al., 2020).

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

We have included data as an attachment.

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CRediT authorship contribution statement

PD conceived and planned the study. SW carried out the literature search and meta-analysis. EB carried out the on-farm studies of insect biodiversity. PD performed all statistical analyses. PD, SW, EB and DC contributed to writing the manuscript.

Author contributions

All authors contributed to conceiving and writing the manuscript.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2022.108254.

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