



Sown mini-meadows increase pollinator diversity in gardens

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Abstract

Habitat loss and fragmentation are considered the foremost threats in pollinator decline, and in England and Wales, 97% of wildflower meadows were lost by 1984. The value of creating flower-rich margins in agricultural environments is established, yet there is growing potential to support pollinator populations in urban landscapes. We used citizen science to investigate the effectiveness of small 4m² sown wildflower ‘mini-meadows’ in UK gardens and allotments in recruiting beneficial insects. Participants were allocated one of three treatment groups: Mix 1 (commercially available ‘meadow mix’); Mix 2 (formulated based on existing literature on pollinator foraging preferences); or Control (no additional wildflowers). All participants conducted insect sampling over two years using standardised pan and sticky trap methods May–August. Samples were returned for identification by trained specialists. Mini-meadows provided resource-rich habitats, increasing wild bee richness and supporting on average 111% more bumblebees, 87% more solitary bees and 85% more solitary wasps in the year following seed-sowing, compared to Control plots. The wildflower mixes were also taxon-specific in their attractiveness. Mix 1 attracted more solitary bees and bumblebees, whereas Mix 2 attracted more solitary wasps. There was no significant difference in the abundance of hoverflies between treatments. Higher abundance of solitary wasps and bees caught amongst the mini-meadow was perhaps due to shorter foraging ranges.

Implications for insect conservation Domestic gardens and allotments provide huge potential habitat for pollinators, and small-scale floral enhancements can attract more beneficial insects in fragmented urban landscapes, supporting urban biodiversity, pollination services and biological control.

Keywords Solitary wasp · Wildflower · Citizen science · Bumblebee · Solitary bee · Wildlife gardening

Introduction

Expanding urbanisation is a significant driver of habitat loss and fragmentation, with 55% of the global human population now living in urban environments (Vié et al. 2009; UN 2019). Habitat loss and fragmentation are considered one of the foremost threats to the decline of pollinators, reducing the availability of essential pollen, nectar and refuge (Goulson et al. 2015). In England and Wales, 97% of lowland wildflower meadows were lost between 1930 and 1984 (Fuller 1987).

The availability of floral resources directly influences bee abundance (Roulston and Goodell 2011), and so most agri-environment schemes (AES) implemented across Europe include strategies to boost the number of flowers, such as

sowing flower-rich margins to provide habitat and forage for pollinators in agricultural landscapes (DEFRA 2020). There is evidence that such schemes do increase both pollinator abundance (eg. Carreck and Williams 2002; Carvell et al. 2007) and the abundance of some natural predators of pests (Tschumi et al. 2015). Like wild bees, solitary wasps depend on plants for pollen, nectar, nesting and overwintering sites during their life cycle (Tschardt et al. 1998). Sown wildflowers can attract parasitoid wasps in agricultural landscapes, thereby enhancing natural pest control (Hoffmann et al. 2018). Recent studies emphasise considering bee and non-bee species when designing floral mixes (Howlett et al. 2021).

Compared to agricultural landscapes, fewer studies have been conducted in gardens on the link between additional floral resources and pollinator abundance, yet they contribute considerable green space to urban areas. Gardens comprise an estimated 24–36% of the area of UK cities (Baldock et al. 2019) covering an area of 400,000 ha (The Wildlife

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Trust 2021). Similarly, gardens account for 36% of urban space in a New Zealand city (Mathieu et al. 2007), although this can vary greatly, with gardens accounting for just 16% of urban space in Stockholm, Sweden (Colding et al. 2006). Likewise, in developing countries, gardens also contribute essential green space to cities. Private garden patios in León, Nicaragua for example, account for 86% of the city's green space (González-García and Sal 2008).

Gardens differ considerably from rural farmland landscapes, with high numbers of non-native plants, and often substantial areas of impervious surfaces (Matteson and Langellotto 2011) and have less area available for sowing of wildflower patches. Given the considerable area of potential habitat for pollinators represented by gardens collectively in Europe and beyond, more research is needed to understand the value and effects of enhancing floral resources in these settings.

Although urbanisation is often regarded as having negative impacts on biodiversity in general, a considerable diversity of bees can be found in cities and urban areas (Fortel et al. 2014; Lanner et al. 2019), particularly in gardens and allotments (Baldock et al. 2019). For pollinators to thrive in urban green spaces there must be sufficient nesting/breeding opportunities and an adequate supply of foraging plants (Goulson et al. 2015; Splitt et al. 2021). The planting of additional floral resources is considered to positively impact pollinator abundance and richness in gardens (Pawelek et al. 2009; Salisbury et al. 2015; Garbuzov and Ratnieks 2014), although there are exceptions (eg. Matteson and Langellotto 2011).

In agricultural landscapes, providing particular plant species rather than increasing overall plant richness is most effective in increasing pollinator abundance (Warzecha et al. 2018). Similarly, in a study of flowering species specifically grown in discrete patches, 18 out of 40 bee-friendly wildflowers provided forage for 100% of observed bee species (Nichols et al. 2019), and garden flowers can vary approximately 100-fold in their attractiveness to insects (Garbuzov and Ratnieks 2014), suggesting selectivity over plant species is key to increasing pollinator abundance (Nichols et al. 2019). Different types of plants are attractive to different pollinator groups, for example, bumblebees are commonly attracted to long-corolla flowers and parasitoid wasps and hoverflies to short-corolla flowers (Campbell et al. 2012). Additionally, native flowers are considered to have positive effects on bee richness and abundance in gardens (Pardee and Philpott 2014; Rollins and Goulson 2019). Strategies to support pollinators recommend planting pollen and nectar-rich plants in green spaces (eg. Royal Horticultural Society 2021, 'Plants For Pollinators'). Ready-to-sow wildflower mixes targeting pollinators are readily available, although to the best of our knowledge, there are no previous published studies on how successful these mixes are at increasing the

abundance of insects and richness of bee species in domestic gardens or allotments.

Citizen science (also known as 'community science') is used in multiple disciplines and the potential for monitoring is recognised by the United Nations Sustainable Development Goals (Fraisl et al. 2020). Citizen science projects focusing on bees have contributed valuable data (eg. Birkin and Goulson 2015) gaining data on a temporal and spatial scale that would otherwise be difficult to achieve. However, bee-based citizen science is biased towards social species, with fewer projects on solitary bees (Koffler et al. 2021). In our study, we used citizen science as a novel and pragmatic approach to access private gardens to survey insects.

The 'Sow Wild!' project focused on the effectiveness of sown mini-meadows in UK domestic gardens and allotments, addressing the following questions: i) Does the creation of a mini-meadow increase the abundance of 'beneficial insects' (pollinators and natural enemies of pests) and richness of bee species. ii) Do wildflower mixes differ in their success in recruiting different groups of beneficial insects. iii) Does a mini-meadow have a positive 'spillover' effect on pollinator abundance throughout the garden or allotment.

Methods

Citizen scientist recruitment for 'Sow Wild!'

Participants were recruited in 2015 through social media, via allotment societies and members of 'The Buzz Club' (a citizen science charity based at the University of Sussex <https://www.thebuzzclub.uk/>). Expression of interest was obtained via an online survey, with the basic requirements being that participants had a garden or allotment (hereafter 'site') of at least 20m² and space of 2×2m to establish a 'mini-meadow' wildflower patch. Participants meeting these requirements then completed a second survey asking detailed information on their site management. A private Facebook group was created to encourage engagement.

One hundred and fifty participants were randomly split into three groups of 50 participants, receiving Mix 1 seeds, Mix 2 seeds, or Control. The control group did not receive any seed mixes but still conducted insect sampling in their garden. Experiments were conducted in 2016 (Year 1) and 2017 (Year 2).

Wildflower mixes

Mix 1 (Table 1) is based on a mix recommended under the UK's Countryside Stewardship Scheme for the establishment of flower-rich plots under its AES, a general-purpose 'Meadow Mix' (Emorsgate EM3 (2016 composition), Emorsgate Seeds, UK). We also added *Papaver rhoeas* and

Table 1 Mix 1 and Mix 2 flowering plant and grass composition. Species labelled * (Mix 1) or ** (Mix 2) did not flower in any site during the study. Letters refer to whether species are annual (a), biennial (b) or perennial (p)*Mix 1 and Mix 2 common flowering plants*

Centaurea cyanus (a)
Centaurea nigra (p)
Centaurea scabiosa (p)**
Daucus carota (b)
Leontodon hispidus (p)
Leucanthemum vulgare (p)
Lotus corniculatus (p)
Papaver rhoeas (a)
Ranunculus acris (p)

Additional flowering plants

Mix 1	Mix 2
<i>Achillea millefolium</i> (p)	<i>Alliaria petiolata</i> (b)
<i>Betonica officinalis</i> (p)	<i>Barbarea vulgaris</i> (b/p)
<i>Filipendula ulmaria</i> (p)*	<i>Campanula glomerata</i> (p)**
<i>Galium album</i> (p)	<i>Echium vulgare</i> (b)
<i>Galium verum</i> (p)	<i>Hypochaeris radicata</i> (p)
<i>Origanum vulgare</i> (p)	<i>Knautia arvensis</i> (p)**
<i>Plantago media</i> (p)	<i>Matricaria chamomilla</i> (a)
<i>Primula veris</i> (p)	<i>Onobrychis vicifolia</i> (p)**
<i>Prunella vulgaris</i> (p)	<i>Reseda lutea</i> (b/p)
<i>Rhinanthus minor</i> (a)	<i>Scabiosa columbaria</i> (p)**
<i>Rumex acetosa</i> (p)	<i>Scorzoneroideis autumnalis</i> (p)**
<i>Sanguisorba minor</i> (p)	
<i>Silene dioica</i> (p)	
<i>Lychnis flos-cuculi</i> (p)	
<i>Trifolium pratense</i> (p)	
<i>Vicia cracca</i> (p)	

Grasses

Mix 1	Mix 2
<i>Agrostis capillaris</i>	<i>Cynosurus cristatus</i>
<i>Cynosurus cristatus</i>	<i>Festuca rubra ssp commutata</i>
<i>Festuca rubra</i>	<i>Festuca rubra ssp juncea</i>
<i>Phleum bertolonii</i>	<i>Poa pratensis</i>

Centaurea cyanus to the mix, to provide additional floral cover in the first year, and reduce weed competition. Mix 2 (Table 1) was formulated based on existing literature and personal communications with Brown, R, and Wood, T.J, identifying flowers to attract a range of pollinator species and providing flowering cover across the season. Mix 2 was formed mostly of perennials as they produce more pollen and nectar than annual flowers (Hicks et al. 2016), create more overwinter nesting capacity for insects (Ganser et al. 2019) and last multiple seasons. Species commonly included in commercial mixes include *Centaurea cyanus*, *Leucanthemum vulgare*, *Centaurea nigra*, *Daucus carota*, *Lotus corniculatus*, *Silene dioica*, and *Trifolium pratense* (Hicks et al. 2016 and references therein) and these were included in both mixes.

Year 1 materials and methodology

Participants received a project pack, including: 16 g wild-flower seeds (according to group allocation), specimen jars (Medline 200 ml Polypropylene Container, Rapid Electronics, UK), pan traps, printed instructions, data collection workbook (supplementary material S1) and ID guides. Pan traps were spray painted by hand, and a set consisted of three 750 ml takeaway-style plastic food containers (Go Packaging Products, UK), one white, one pink (Rust-Oleum spray paint Direct to Plastic White and Rust-Oleum Painters Touch Berry Pink Gloss, Rust-Oleum Corporation, US), and one blue (PlastiKote Pacific Blue Gloss: PlastiKote, Valspar, US).

In April, Mix 1 and Mix 2 group participants were instructed to sow their wildflower seeds at 4 g/m² to create a mini-meadow (supplementary material S1). Insect sampling using pan traps took place during the first week of the months May–August, over a dry and sunny 48-h period. Mix 1 and Mix 2 were instructed to place one set of pan traps side by side in the middle of the mini-meadow, and a second set in a designated area 10 m away from the mini-meadow and not amongst garden flowers. Control group participants were instructed to place a single set of pan traps in their site, away from existing garden flowers. Pan traps were ¾ filled with water and a squeeze of lightly fragranced washing-up liquid ('Ecover' was recommended: Ecover, Malle, Belgium), and left undisturbed for 48 h. Specimens were collected in labelled jars of clear distilled household vinegar.

Each month, all participants were instructed to complete the workbook, identifying insects collected to group: bumblebee, honeybee, solitary bee, wasp, hoverfly, butterfly, moth, other fly, other insects. Mix 1 and Mix 2 groups listed the flowering species appearing in the mini-meadow. Participants in all groups were instructed to list and estimate other plants species flowering in the rest of their garden or allotment using the following scale: 1–10, 11–25, 26–100, 101–200, 201–1000, 1001–5000, 5000+ plants (Carvell et al. 2007). Participants took photos each month of the mini-meadow and/or site.

Year 2 materials and methodology

Sampling commenced as in Year 1, with some adaptations based on participant feedback designed to improve the insect sampling methods. Yellow sticky insect traps (Gardening Naturally, UK) were co-located with pan traps, attached to a bamboo cane elevated ½ metre in situ for 2 weeks, then labelled and covered in clingfilm. A fourth yellow pan trap (Rust-Oleum Painters Touch Sun Yellow Gloss: Rust-Oleum Corporation, US) was also added to the set. A large asterisk was drawn in thick permanent black marker pen (Sharpie, Sanford L.P, US) on the inside of all pan traps to act as a 'nectar guide'. Participants were explicitly asked to remove slugs, snails, butterflies and moths from samples as in Year 1 these were found to partially dissolve in vinegar and made insect identification difficult.

Identification of samples

Insect sample pots, sticky traps and workbook recording sheets were returned via post, and photographs returned digitally. Pan trap and sticky trap insects were sorted by researchers in the laboratory to broad insect group, with all pan trap bees and hoverflies identified to species level. Hereafter 'solitary bees' refers to all non-corbiculate bees (*i.e.* all bees except bumblebees and honeybees), and 'wild

bees' refers to both solitary bees and bumblebees (*i.e.* all bees except honeybees).

Data analysis

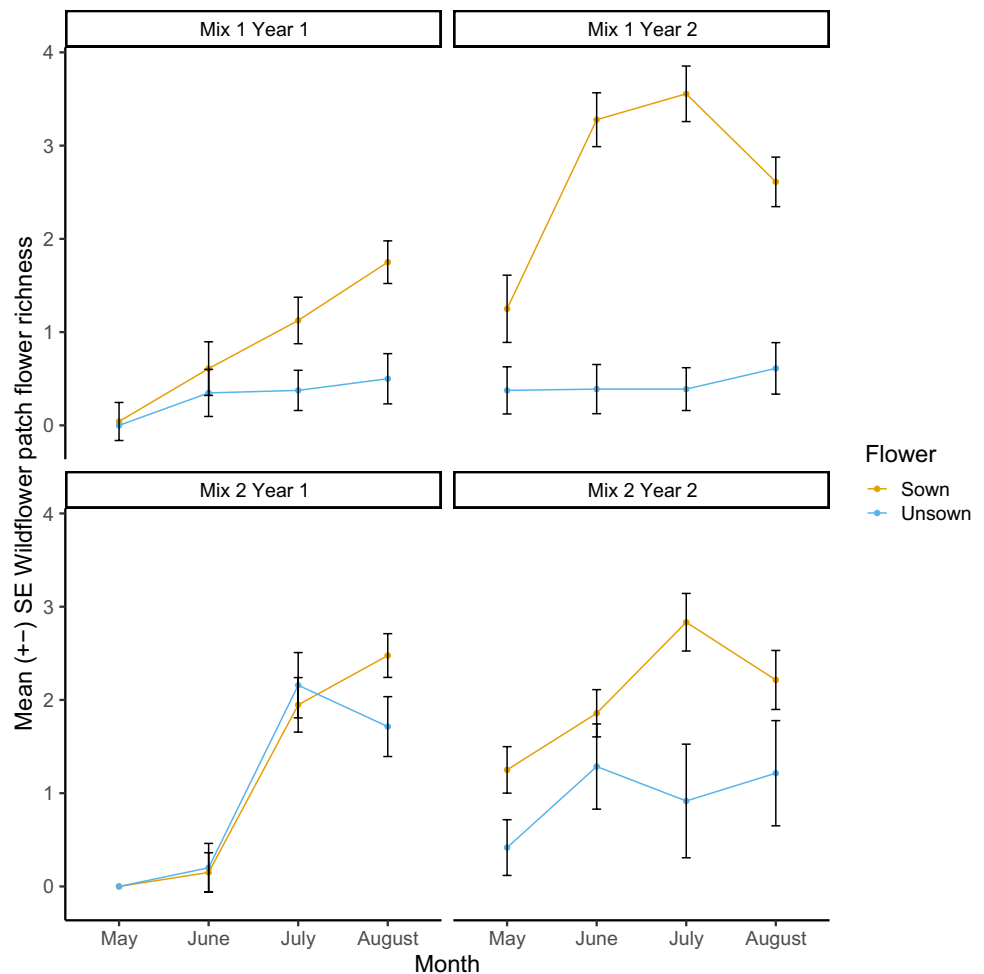
Data analysis was conducted in R (R core team 2020). A Shapiro–Wilk normality test was conducted to test for parametric data. Generalised Linear Mixed Models (GLMMs) were built using *lme4* package, zero-inflated models were built using *glmmTMB* package. Pan trap data for Year 1 and Year 2 were analysed separately due to changes in sampling methods and participant drop-out in Year 2. Models of best fit were chosen based on diagnostic residual plots and AIC values. ANOVAs were performed by comparing full and reduced models and reported as chi-square values. Tukey's Honest Significant Difference test was used to compare between Mix 1 and Mix 2.

A Shannon Diversity Index of other garden flowers present in the rest of the site was calculated per site per month, using richness and abundance data (mid-point of flowering plant count scale) provided by participants. 'Total insect abundance' includes: solitary bees, bumblebees, honeybees, hoverflies, social wasps, solitary wasps and 'other' flies. Analysis of 'bee richness' includes species of solitary bees, bumblebees and honeybee. Hoverfly richness could not be analysed, as too few hoverflies were sampled over the two years.

To test whether the creation of mini-meadows increases the abundance of beneficial insects, 'Total insect abundance' was used as a response variable. The total insect abundance counts from pan traps set inside the mini-meadows (Mix 1 and Mix 2 data combined) was compared to counts from pan traps in Control sites, irrespective of mix. Trap placement (inside mini-meadow *vs.* Control), month and Shannon Diversity Index of other garden flowers were predictor variables. Site ID was allocated as a unique identifier and used as a random variable. GLMM with negative binomial family was fitted for Year 2 pan trap data, whereas a GLMM with Poisson family was fitted for Year 1 pan trap and Year 2 yellow sticky trap data. To test the effects of the mini-meadow on the abundance specific insect groups, these were considered separately (bumblebees, solitary bees, hoverflies and solitary wasps), as was bee species richness (all with GLMMs with zero-inflated negative binomial distribution).

Secondly, we wanted to test if the wildflower mixes affected the response variable 'Total insect abundance'. A GLMM (poisson) was used with treatment (Mix 1, Mix 2, Control), month, and Shannon Diversity Index of other garden flowers as predictor variables, and site ID as a random variable. Insect group abundance was considered separately, as was bee species richness (GLMMs with zero-inflated negative binomial distribution).

Fig. 1 Mean (\pm SE) Richness of sown and unsown wildflower in Mix 1 and Mix 2 in Year1 and Year 2



To investigate how localised the effect of the mini-meadows on pollinator abundance, ‘Total insect abundance’ was compared between pan traps placed directly within the mini-meadow, with those placed 10 m away. Data from Mix 1 and Mix 2 were combined for this analysis. A GLMM (negative binomial) was modelled with trap placement (inside mini-meadow *vs.* 10 m away), month, and Shannon Diversity Index of other garden flowers were included as predictor variables, and site ID as a random variable. Again, insect group abundance was also analysed separately, as was bee species richness (GLMMs with zero-inflated negative binomial distribution).

Lastly, we used rarefaction analysis to explore the diversity of bee species of Mix 1, Mix 2 and Control sites, allowing comparison of unequal sample sizes (Hsieh et al. 2016). Rarefaction and extrapolation curves were created using three diversity orders of Hill numbers: species richness ($q=0$), Shannon diversity ($q=1$) and Simpson diversity ($q=2$) with 95% confidence intervals, all computed in the iNEXT package (Hsieh et al. 2016). Diversity measures differ significantly at $p \leq 0.05$ if the 95% confidence intervals (CI) do not overlap (Colwell et al. 2012).

Results

Mini-meadow establishment

Sown wildflower species richness increased annually from Year 1 to Year 2, when considering all the floral data collected across the four sampling months and both mixes (Mean \pm SE: 1 ± 0.11 , to 2.43 ± 0.11 respectively) as was expected with the establishment and flowering of more biennial and perennial species in the second year. Mix 1 saw a greater annual increase in sown richness (number of sown flowering species) on average (Mean \pm SE: 0.88 ± 0.14 to 2.71 ± 0.16 respectively) compared to Mix 2 (Mean \pm SE: 1.14 ± 0.17 to 2.04 ± 0.15 respectively). Mix 2 patches had a higher richness of unsown flowers (species not included in the seed mix) in both years of study, on average (Mean \pm SE: Year 1: 1 ± 0.19 , Year 2: 0.98 ± 0.26) compared to Mix 1 (Mean \pm SE: Year 1: 0.31 ± 0.13 , Year 2: 0.44 ± 0.13). In Mix 1, 24 (96%) of the wildflower species contained in the mix flowered during the study in at least one site, compared to 19 (68%) for Mix 2 (Table 1). Seasonal changes are seen

Table 2 Most observed wild bee species in UK domestic gardens and allotments captured using pan traps, combining Year 1 and 2 data

Wild bee species	Count
<i>Bombus terrestris</i> agg	169
<i>Lasioglossum leucopus</i>	101
<i>Lasioglossum smeathmanellum</i>	79
<i>Lasioglossum morio</i>	74
<i>Bombus pascuorum</i>	56
<i>Bombus pratorum</i>	43
<i>Bombus hortorum</i>	33
<i>Halictus tumulorum</i>	31
<i>Lasioglossum calceatum</i>	26
<i>Hylaeus hyalinatus</i>	25
<i>Lasioglossum albipes</i>	24
<i>Lasioglossum minutissimum</i>	23
<i>Bombus lapidarius</i>	22
<i>Osmia bicornis</i>	22
<i>Bombus hypnorum</i>	20
<i>Lasioglossum pauxillum</i>	18
<i>Andrena haemorrhoa</i>	17
<i>Lasioglossum cupromicans</i>	13
<i>Megachile centuncularis</i>	12
<i>Andrena bicolor</i>	10
<i>Anthidium manicatum</i>	10

within both mixes, with flower richness peaking in July in Year 2 for both mixes (Fig. 1).

Insect abundance in gardens and allotments

Over two years, a total of 454 bumblebees, 218 hoverflies, 877 solitary bees, 176 honeybees, 4,443 solitary wasps and 28,270 ‘other’ flies were sampled. Sixty-six species of wild bee were identified to species level from pan trap samples, spanning 14 genera and including ten species of bumblebee. The most abundant wild bee species are listed in Table 2. Twenty-two species of hoverfly were identified over 16 genera: the most abundant *Helophilus pendulus*, accounted for 38% of hoverflies identified to species, followed by *Episyrphus balteatus* at 11%. (Full species list for bees and hoverflies available in supplementary material S2).

Table 3 Distribution of treatment group participants in Year 1 and 2. Percentage of sites located in urban locations (versus rural) and average size of sites in each treatment for Years 1 and 2

Treatment	Year 1 participants	Year 2 participants	Urban sites Year 1/Year 2	Site size (Avg.) Year 1/Year 2
Mix 1	37% (N=25)	38% (N=18)	92/94%	217/238m ²
Mix 2	34% (N=23)	31% (N=15)	96/100%	263/276m ²
Control	29% (N=20)	31% (N=15)	85/87%	228/235m ²

Citizen scientist participation

Out of the initial 150 participants, 68 (45%) returned samples in Year 1. In Year 2, 48 (32%) returned pan trap samples and 46 (31%) returned sticky traps samples (deployed in Year 2 only). Participants that submitted data in both years of the study were evenly distributed across treatment groups (Table 3), meaning drop-out rates were likely not affected by the treatment group a participant was assigned to. The average size of the site was 236m² in Year 1 and 249m² in Year 2 (Table 3). The majority of sites were in urban locations [Table 3; based on Rural Urban Classification (DEFRA 2021)].

Forty-three (63%) participants submitted photographs of the mini-meadow and/or site in Year 1, dropping to twenty-three (48%) in Year 2 (Fig. 2). However, photographs were non-standardised, and therefore abundance of individual flower species could not be discerned, especially smaller species. Photographs showed that in Year 1, when present, *Centaurea cyanus* appeared to dominate the flower patches, followed by *Papaver rhoeas*, *Silene dioica* and *Leucanthemum vulgare*. In Year 2, *Leucanthemum vulgare*, *Daucus carota*, *Ranunculus acris*, *Silene dioica*, the knapweeds (*Centaurea* spp.) and the dandelion-like flowers (e.g. hawkbits) dominated when present.

Do mini-meadows increase the abundance of beneficial insects?

When the insect abundance data from Mix 1 and 2 were combined and compared against the Control (with no mini-meadow), there was no significant difference in total insect abundance (all solitary bees, bumblebees, honeybees, hoverflies, social wasps, solitary wasps and ‘other’ flies) in either Year 1 or Year 2 pan traps or Year 2 sticky traps (Table 4). However, in Year 2, when flowering plant species richness was highest, significantly more bumblebees were caught in pan traps, and significantly more solitary bees and solitary wasps were caught using sticky traps, in sites with a mini-meadow compared to sites without (Fig. 3; Table 4). There was no significant difference in the abundance of hoverflies between sites with or without a mini-meadow for any year or sampling method used (Table 4). Bee richness did not differ between sites with a mini-meadow compared to Control in

Fig. 2 Example of the mini-meadow photographs submitted by participants. Clockwise from top left: Year 1 Mix 1; Year 1 Mix 2; Year 2 Mix 2; Year 2 Mix 1 (photos courtesy of Amanda James, Anne Macarthur, name withheld & Judith Gray respectively)



Year 1. In Year 2 however, mini-meadow had significantly more bee species than Control (Table 4).

We considered the difference between the mean observed counts of bumblebees, solitary bees and solitary wasps, comparing those captured at sites with a mini-meadow compared to Control, across all months and locations. In Year 1, sites with a mini-meadow had 109% more bumblebees, 24% more solitary bees, and 126% more solitary wasps compared to sites without. In Year 2 sites with a mini-meadow supported 111% more bumblebees, 87% more solitary bees and 85% more solitary wasps than Control.

Do different mixes recruit different beneficial insect groups?

In Year 1, in total there were significantly more insects caught in the Mix 1 mini-meadows compared to Mix 2 (Table 4). This is largely driven by the high number of flies caught in both Mix 1 mini-meadows (mean \pm SE: 43.2 ± 0.65) and control sites (43.3 ± 0.74), compared to Mix 2 mini-meadows (28 ± 0.43). In Year 2 there was no significant difference in overall insect abundance between the three treatments (Mix 1, Mix 2, Control), for either pan trap or sticky trap caught insects. (Table 4).

In both Year 1 and Year 2, there were significantly more pan trap-captured solitary bees in Mix 1 compared to Mix 2. Furthermore, in Year 2, sticky traps caught more solitary bees in both Mix 1 and Mix 2, compared to Control (Table 4; Fig. 4). In Year 2 pan traps, Mix 1 caught significantly more bumblebees than Control (Table 4; Fig. 4). There was no difference in bumblebee abundance between the three treatments in Year 1, or sticky traps in Year 2.

There was a significant difference between treatments in the abundance of solitary wasps caught using sticky traps, with post hoc tests indicating that there were significantly more solitary wasps captured in Mix 2 mini-meadows than Control (Table 4; Fig. 5). There was no significant difference in the abundance of solitary wasps in pan traps between the different mixes in Year 1 or Year 2. There was no significant difference in the abundance of hoverflies for either year or sampling method.

In both Year 1 and 2, Mix 1 mini-meadows had significantly higher bee species richness than both Mix 2 or Control (Fig. 6).

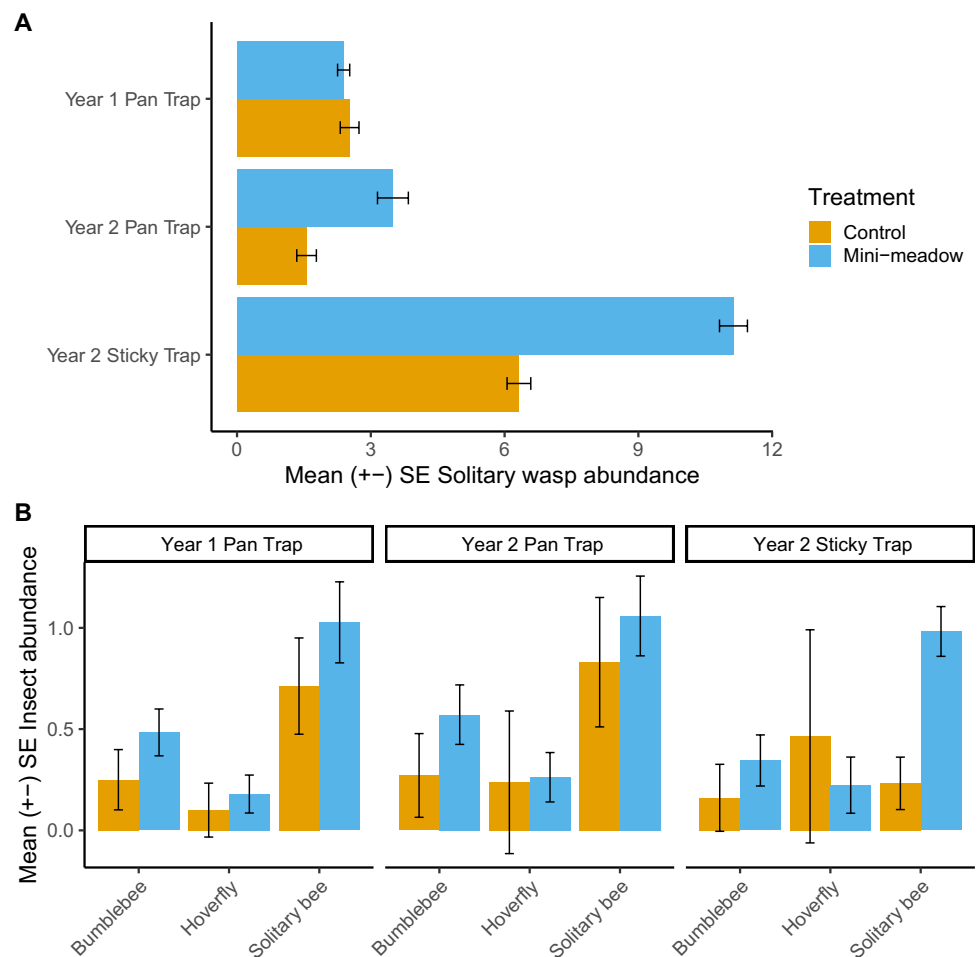
In Year 1, rarefaction analysis across the three diversity measures (species richness, Shannon diversity, Simpson diversity) showed little difference in the bee species composition of the sites according to treatment, with 95% CI overlapping (Fig. 7A). In Year 2, however, rarefaction analysis

Table 4 Insect abundance and bee richness of pan traps in Year 1 (PT Y1) and Year 2 (PT Y2) and yellow sticky traps Year 2 (YT Y2)

Abundance	Method	GLMM by treatment (i)		Sign	Control	Mix 1	Mix 2	GLMM mini-meadow (ii)		Sign	All mix	
		χ^2	df					χ^2	df			
Total insect	PT Y1	6.63	2	0.04	47.2±0.72 (ab)	48.2±0.64 (b)	29.4±0.43 (a)	0.99	1	0.32	NS	39.2±0.42
Solitary wasp	PT Y1	2.76	2	0.25	2.52±0.21	2.79±0.21	1.96±0.16	0.22	1	0.64	NS	2.39±0.14
Solitary bee	PT Y1	9.89	2	0.01	0.71±0.24 (ab)	1.59±0.29 (b)	0.38±0.16 (a)	0.60	1	0.44	NS	1.03±0.2
Bumblebee	PT Y1	3.22	2	0.20	0.25±0.15	0.52±0.14	0.44±0.19	2.01	1	0.16	NS	0.48±0.12
Hoverfly	PT Y1	4.00	2	0.14	0.1±0.13	0.24±0.13	0.12±0.14	1.30	1	0.25	NS	0.18±0.09
Total insect	PT Y2	1.06	2	0.59	31.4±0.56	32.9±0.49	30.6±0.71	<0.01	1	0.95	NS	31.9±0.41
Solitary wasp	PT Y2	4.91	2	0.09	1.56±0.22	2.58±0.22	4.75±0.63	3.13	1	0.08	NS	3.52±0.34
Solitary bee	PT Y2	9.53	2	<0.01	0.83±0.32 (ab)	1.61±0.26 (b)	0.28±0.14 (a)	1.77	1	0.18	NS	1.03±0.2
Bumblebee	PT Y2	6.54	2	0.04	0.27±0.21 (b)	0.58±0.16 (a)	0.55±0.27 (ab)	6.29	1	0.01	*	0.57±0.15
Hoverfly	PT Y2	1.29	2	0.53	0.24±0.35	0.26±0.17	0.27±0.18	1.91	1	0.17	NS	0.26±0.12
Total insect	YT Y2	3.72	2	0.16	20.2±0.4	22.9±0.34	28.1±0.48	1.38	1	0.24	NS	25±0.29
Solitary wasp	YT Y2	8.44	2	0.02	6.32±0.27 (a)	9.7±0.28 (ab)	13.2±0.61 (b)	7.12	1	<0.01	**	11.1±0.31
Solitary bee	YT Y2	13.63	2	<0.01	0.23±0.13 (b)	1.04±0.16 (a)	0.89±0.18 (a)	12.78	1	<0.001	***	0.98±0.12
Bumblebee	YT Y2	2.21	2	0.33	0.16±0.17	0.37±0.17	0.3±0.18	2.21	1	0.14	NS	0.35±0.13
Hoverfly	YT Y2	0.07	2	0.97	0.46±0.53	0.21±0.17	0.24±0.23	2e-04	1	0.99	NS	0.22±0.14
Richness (iii)												
Bee richness	PT Y1	12.37	2	<0.01	0.73±0.15 (a)	1.34±0.16 (b)	0.71±0.18 (a)	1.31	1	0.25	NS	1.01±0.12
Bee richness	PT Y2	12.38	2	<0.01	0.7±0.2 (a)	1.54±0.14 (b)	0.6±0.16 (a)	4.07	1	0.04	*	1.19±0.11

GLMM ANOVA results for effects of (i) Treatment (Mix 1, Mix 2 and Control) on the abundance of all insects, solitary wasps, solitary bees and bumblebees, (ii) mini-meadow (all mixes) versus Control and (iii) effects of treatment on bee richness (solitary bees, bumblebees, honeybees). Presented with mean±standard error, chi-square χ^2 , degrees freedom df, significance NS not significant; *p<0.05; **p<0.01; ***p<0.001) and Tukey's Honest Significant Difference test for comparisons where relevant (designated by letters in bold)

Fig. 3 Mean (\pm SE) abundance of **A** solitary wasps and **B** wild bees and hoverflies found in sites with a mini-meadow (Mix 1&2 combined) and those without (Control) in Year 1 (pan traps only) and Year 2 (pan traps and sticky traps)



across the three diversity measures indicated bee species diversity differed according to treatment (Fig. 7B), with the highest dissimilarity (and species turnover) in species composition in Mix 1 sites. Considering Shannon diversity and Simpson diversity (middle and right panel, Fig. 7B), the species diversity composition of Mix 1 differed significantly from both Mix 2 and Control sites, as the 95% CI are not overlapping.

Analysis was conducted on the effects of the diversity of garden flowers on the abundance of insects and richness of bees (Supplementary material S3). Only bumblebee abundance in Year 2 with sticky traps was predicted by garden flower diversity. Since this is not a primary focus of our study we do not discuss this further.

How localised is the impact of the mini-meadow?

Abundance and diversity of insects were compared within sites between samples collected from pan traps and sticky traps placed directly inside the mini-meadows (combined data from Mix 1 and 2) and samples collected from pan and sticky traps that were placed 10 m away from the meadow.

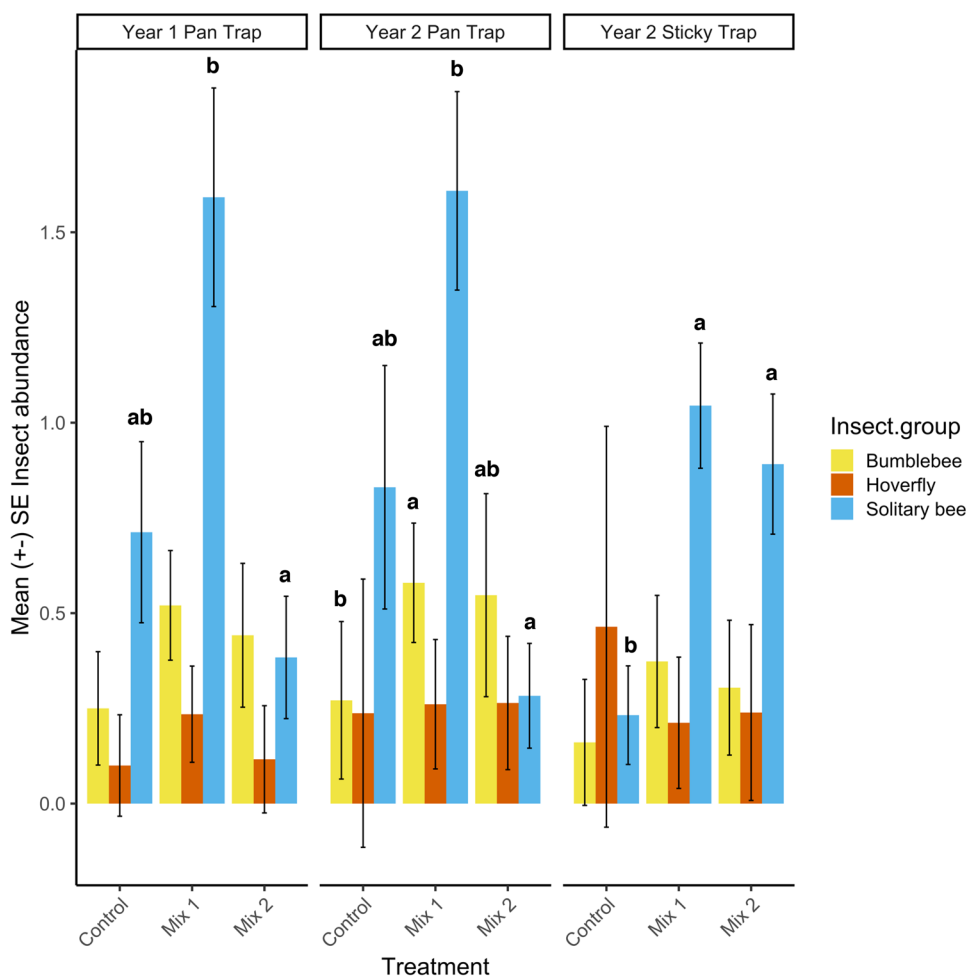
There was no significant difference in the total abundance of insects caught inside the meadows compared to 10 m away for any year or sampling method (Table 5).

However, when comparing the abundance of specific insect groups, in pan trap samples from Year 2, there were significantly more solitary bees and solitary wasps inside the meadows compared to 10 m away, although this pattern wasn't detected in Year 2 sticky traps. Solitary wasp abundance was also significantly higher inside meadows in Year 1 pan trap captures. There was no significant difference in bumblebee or hoverfly abundance inside meadows compared to 10 m away (Table 5). Bee species richness also did not differ inside or 10 m away from the meadow in either year (Table 5).

Discussion

We have demonstrated that sown mini-meadows in domestic gardens and allotments can provide resource-rich habitats for pollinators and solitary wasps, increasing both abundance and richness of wild bee species compared to gardens and

Fig. 4 Mean (\pm SE) abundance of bumblebees, solitary bees and hoverflies sampled in Year 2 (pan traps and sticky traps) and Year 1 (pan traps only) comparing Mix 1, Mix 2 and Control. Letters indicate significant differences in abundances between treatments (Tukey's Honest Significant Difference)



allotments without mini-meadows. Although this study was conducted in the UK, the methodology can be easily replicated in any urban landscape. Significant patterns of abundance of insect groups and bee richness differed between years and sampling methods. This was predominantly due to wildflowers becoming more established in Year 2, and sampling methods differing in their sensitivity of detecting different insect groups. For example, yellow sticky traps were more sensitive to solitary wasp abundance. To obtain more information on wild bee species populations, a combination of sampling techniques are recommended (Templ et al. 2019).

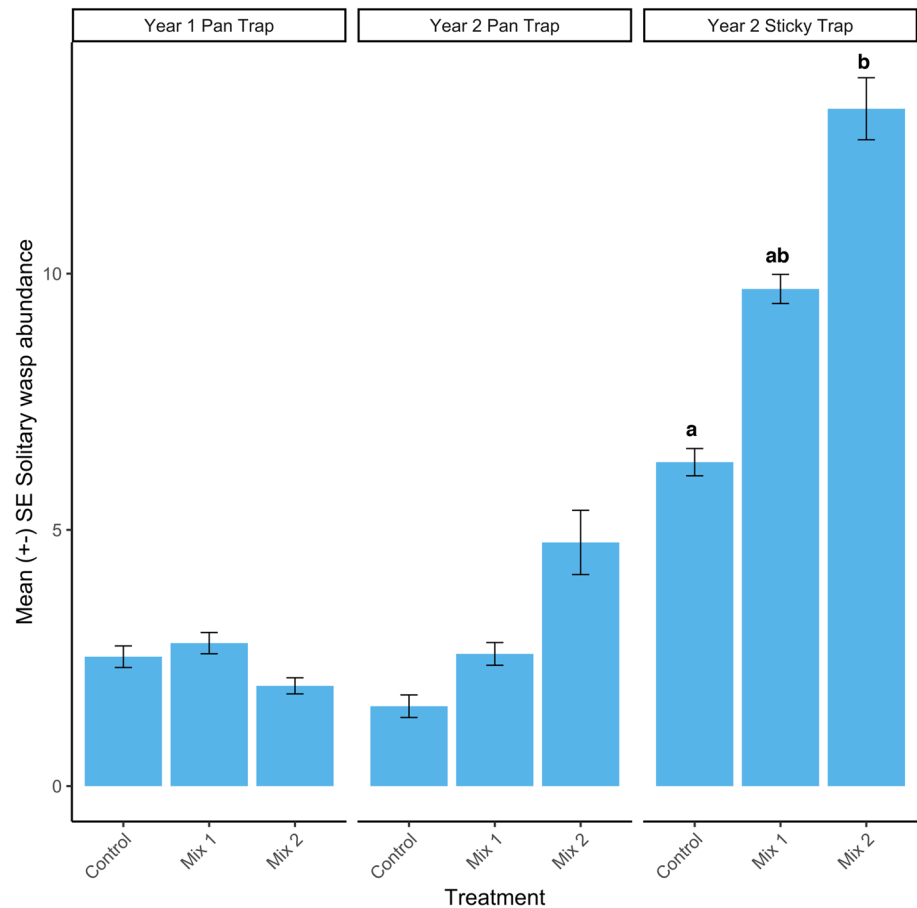
Our results correspond with previous research that found that planting flowers in gardens increases bee richness (Pawelek et al. 2009; Salisbury et al. 2015). Though Matteson and Langellotto (2011) concluded that floral additions in New York community gardens do not increase pollinator richness, our sites were notably smaller on average (Year 1 236m²/Year 2 249m², compared to 909m²), and as the authors suggest, additional floral resources placed in a location with a higher baseline abundance of flowers might see negligible impact on pollinator increases. Therefore, there

is potentially a ‘saturation point’, only up to which any floral additions will benefit pollinator numbers (Simao et al. 2018). While garden size can be regarded as a barrier to wildlife gardening (Goddard et al. 2013) we have found that planting a mini-meadow of just 4m² can enhance resources for beneficial insects, with only a small loss of garden space. In fact, more numerous and smaller mini-meadows throughout landscapes may be more beneficial for the recruitment of bees than larger meadows because of such ‘saturation’ effects (Simao et al. 2018).

Our study also recorded quick recruitment of beneficial insects. Sites with mini-meadows supported 109% more bumblebees, 24% more solitary bees, 126% more solitary wasps in Year 1, and 111% more bumblebees, 87% more solitary bees and 85% more solitary wasps in Year 2, when compared to Control sites. Sown wildflowers are known to be utilised by bees relatively quickly, with a previous study stating a quarter of species known from the Munich region were recruited to wildflower strips within one year of sowing (Hofmann and Renner 2020).

Our mini-meadows also supported less well-studied beneficial insects. Non-syrphid flies were the most abundant

Fig. 5 Mean (\pm SE) abundance of solitary wasps sampled in Year 2 (pan traps and sticky traps) and Year 1 (pan traps only) comparing Mix 1, Mix 2 and Control. Letters indicate significant differences in abundances between treatments (Tukey's Honest Significant Difference)



insect group sampled and are increasingly being recognised as key pollinators of food crops (Orford 2015). Though previous studies have found that wildflower patches in urban grasslands increase the abundance of hoverflies (Blackmore and Goulson 2014), surprisingly there was no difference in the hoverfly abundance recorded between sites with and without mini-meadow. The number of hoverflies collected over the entire study was lower than anticipated at a total of 218 insects, so the sampling technique and small sample size may instead be responsible for this result, and also meant that effects on hoverfly species richness could not be investigated further in this study.

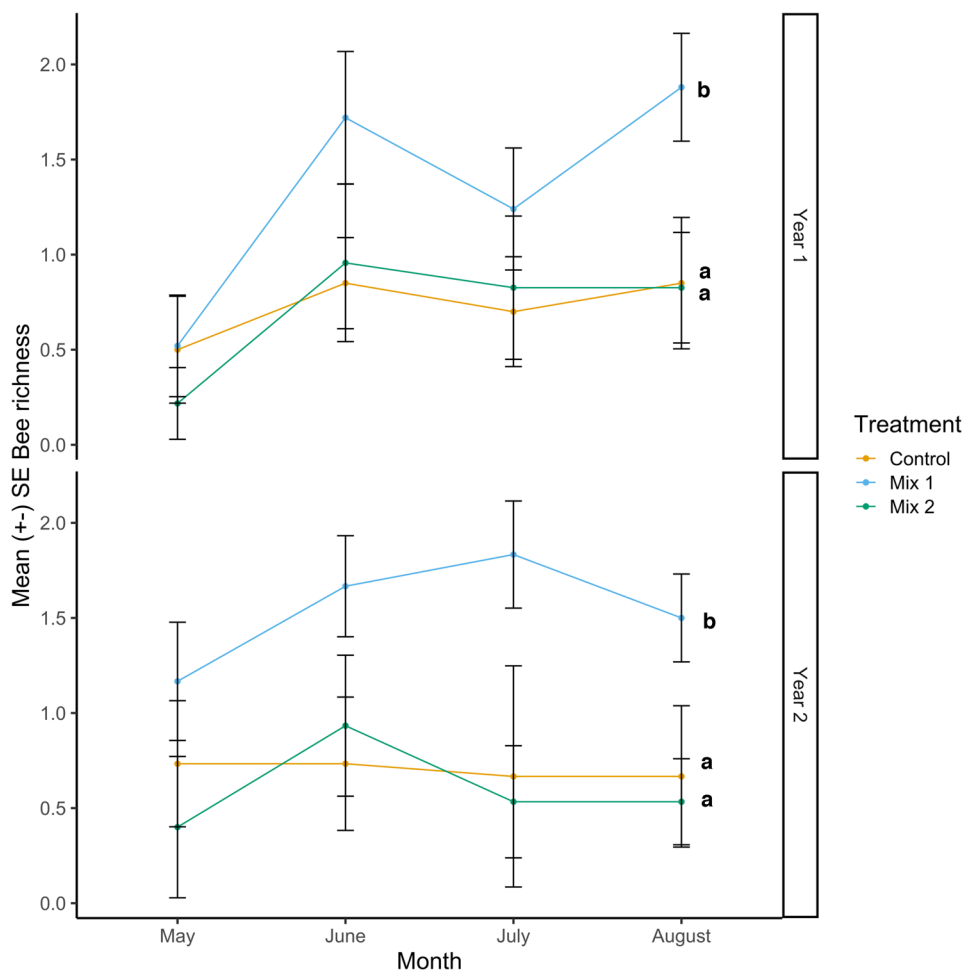
Solitary wasps are a hugely diverse and difficult group of insects to identify, so it was outside the scope of this study to identify this group to species level. However, parasitoid wasps were seemingly numerous in the pan traps (pers. obs.) and identification to species would be an interesting next step. Floral additions provide essential resources to the natural enemies of pest insects as a natural biological control (Araj and Wratten 2015) and sown wildflowers strips are beneficial to ecosystem services by promoting parasitoid wasps in agricultural landscapes

(Hoffmann et al. 2018). Here we show that solitary wasps can also be promoted in domestic gardens and allotments by providing additional floral resources, similar to Bennett and Gratton (2012) who found a positive relationship between parasitoid abundance and floral diversity.

The composition of flowers in the mini-meadows led to recruitment of different taxa. In Years 1 and 2, Mix 1 consistently attracted significantly more individual solitary bees and more species. In Year 2, when the wildflowers were more diverse and established, Mix 1 also attracted significantly more bumblebees, whereas Mix 2 attracted more solitary wasps. Wildflower mixes in agricultural landscapes can be taxon-specific in their attractiveness depending on key plant species in the mix (Warzecha et al. 2018) and we have shown this can also be achieved in domestic gardens and allotments. Identification of such mixes can facilitate conservation efforts (Warzecha et al. 2018).

Certain localised effects on insect abundance were observed in the mini-meadow compared to samples collected 10 m away; both solitary wasps and solitary bees were more abundant inside the meadow. Insects from both these groups

Fig. 6 Mean (\pm SE) bee species richness (bumblebee, solitary bee and honeybee data combined) in Year 1 and 2, comparing Mix 1, Mix 2 and Control. Letters indicate significant differences in richness between treatments (Tukey's Honest Significant Difference)



tend to be smaller in size leading to a more limited foraging range compared to highly-mobile bumblebees which have a foraging range of 1.5 km or more (Osborne et al. 2008). The higher abundance of solitary bees and wasps may also indicate that the wildflowers provide refuge in addition to pollen and nectar. Richness of wild bee species was greater in the gardens which had a mini-meadow compared to those without. This was observed over both years of study and with no localised effects, suggesting the planting of a mini-meadow will increase the overall diversity of wild bees in gardens and allotments through a positive ‘spill-over’ effect.

Flowers of Mix 1 established more successfully, and seasonal flowers appeared over the course of the year, providing a range of different flowering plants for wild bees. Participants did not observe insects directly on flowers, but as the abundance and richness of solitary bees are consistently higher for Mix 1, we expect key species for solitary bees to be present in this mix. Warzecha et al. (2018) identified four plant species that provided resources to 81% of recorded pollinators. Likewise, Nichols et al. (2019) found 14 flower

species accounted for 99.7% of bee visitations. Using direct observation to record such plant-pollinator interactions would be the next step for this study. It would be interesting to determine which sown/unsown species was responsible for the increase in solitary wasp abundance detected in Mix 2, considering the high number of unsown flowers that appeared. Indeed, flowers considered ‘weeds’ can contribute valuable foraging resources; dandelions (*Taraxacum* agg.) produce high quantities of pollen and nectar (Hicks et al. 2016) and enhance biocontrol efficacy by increasing parasitoid longevity and egg load (Araj and Wratten 2015). Studies on biocontrol of pests by enhancing floral resources to enhance natural enemies have often focused on providing just one or a small number of flowering plants; it would be worth investigating a larger range, and the benefits of providing a more diverse flower community (Fiedler et al. 2008).

In this study, citizen scientists made an invaluable contribution, planting and managing wildflowers and completing sampling techniques. However, drop-outs year-on-year may have been non-random. Participants with poorly established

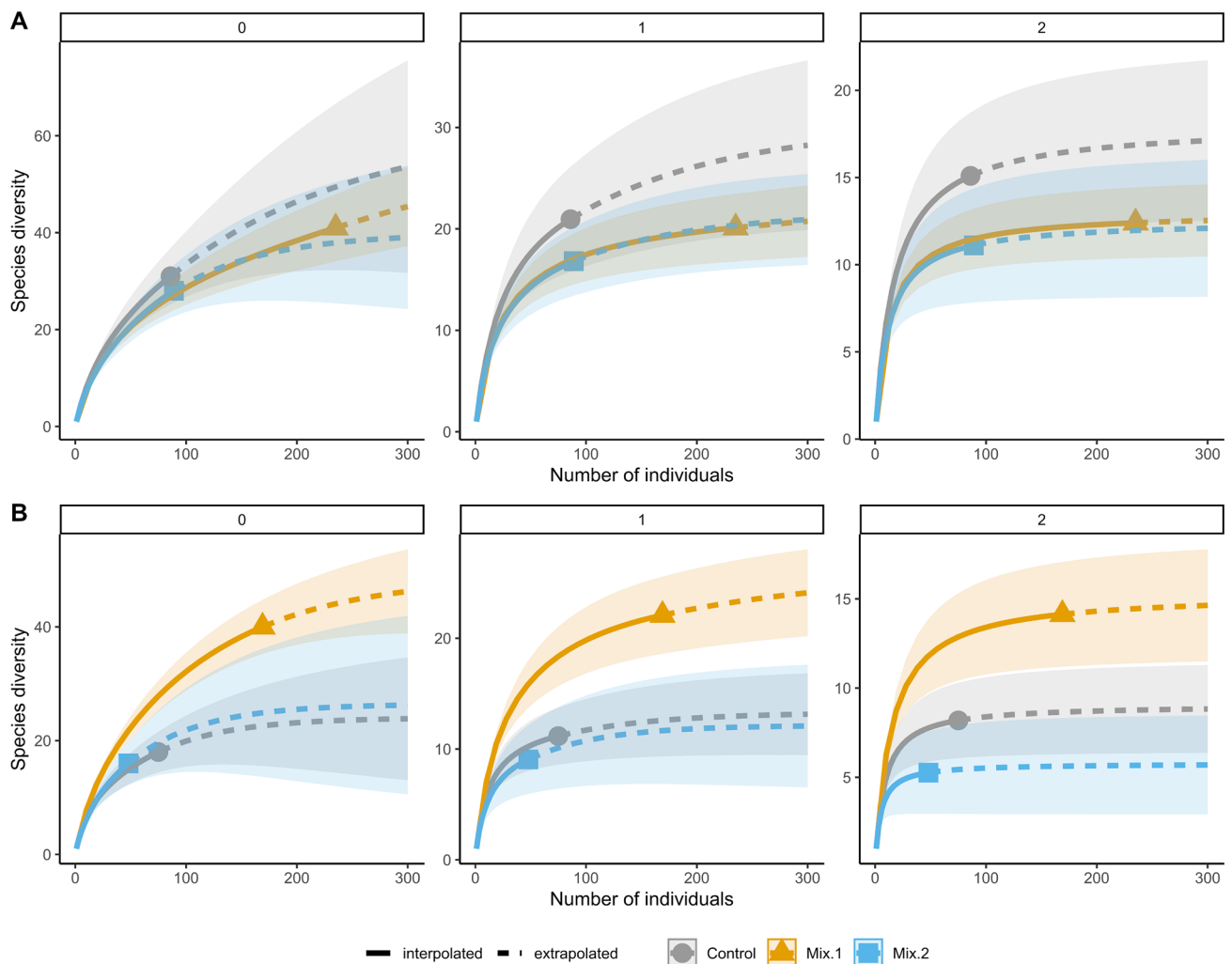


Fig. 7 (A) Year 1 and (B) Year 2 rarefaction curves, showing the expected number of bee species as a function of sampling effort (number of individuals) in the three treatments (Mix 1, Mix 2, Control). Separated by diversity order: $q=0$ (species richness, left panel),

$q=1$ (Shannon diversity, middle panel) and $q=2$ (Simpson diversity, right panel). Solid lines show empirical (interpolated) results, dashed lines show extrapolation. Shaded areas show 95% confidence intervals

wildflowers, or those that caught fewer insects may have left the project, leaving more pollinator-friendly gardens continuing into Year 2. This could potentially bias the increases of insect abundance in Year 2 when comparing to Control gardens.

Horticultural and conservation organisations advise the public on the potential of their gardens to encourage biodiversity, but also of importance is planning policy for new urban developments. A modelling approach by Baldock et al. (2019) found that increasing the area of allotments in cities, and increasing floral abundance in urban greenspaces is beneficial for plant-pollinator interactions and should be considered in urban planning. We support

the notion that gardens and allotments could effectively be included in conservation planning, considering domestic gardens as interconnected habitats and not individual units (Hofmann and Renner 2020). Attracting diverse beneficial insects to gardens and allotments through floral additions has multiple benefits, in addition to enhancing biodiversity. Diverse bee communities enhance urban fruit and vegetable production (Lowenstein et al. 2015) can benefit ecosystem services such as natural pest control and soil protection (Wratten et al. 2012) and enriching a garden has positive impacts on human wellbeing (Fuller et al. 2007).

Table 5 Insect abundance and bee richness in pan traps in Year 1 (PT Y1) and Year 2 (PT Y2) and yellow sticky traps Year 2 (YT Y2). GLMM ANOVA results for comparisons within the mini-meadows (Mix 1 and 2 combined) and 10 m away for abundance of all insects, solitary wasps, solitary bees and bumblebees, and also bee richness (solitary bees, bumblebees, honeybees)

	Method	X^2	df	p=	Sign	Mini-meadow	10 M away
<i>Abundance</i>							
Total insect	PT Y1	0.05	1	0.83	NS	39.2 ± 0.42	39.3 ± 0.43
Solitary wasp	PT Y1	5.15	1	0.02	*	2.39 ± 0.14	1.94 ± 0.14
Solitary bee	PT Y1	0.14	1	0.71	NS	1.03 ± 0.2	0.81 ± 0.12
Bumblebee	PT Y1	0.22	1	0.64	NS	0.48 ± 0.12	0.54 ± 0.14
Hoverfly	PT Y1	1.4	1	0.24	NS	0.18 ± 0.09	0.15 ± 0.12
Total insect	PT Y2	0.002	1	0.97	NS	31.9 ± 0.41	31 ± 0.39
Solitary wasp	PT Y2	4.29	1	0.04	*	3.52 ± 0.34	2.44 ± 0.21
Solitary bee	PT Y2	5.78	1	0.02	*	1.03 ± 0.2	0.65 ± 0.14
Bumblebee	PT Y2	0.54	1	0.46	NS	0.57 ± 0.14	0.48 ± 0.14
Hoverfly	PT Y2	2.52	1	0.11	NS	0.26 ± 0.12	0.12 ± 0.1
Total insect	YT Y2	2.38	1	0.12	NS	25 ± 0.29	22.3 ± 0.26
Solitary wasp	YT Y2	1.98	1	0.16	NS	11.1 ± 0.31	8.89 ± 0.19
Solitary bee	YT Y2	0.37	1	0.54	NS	0.98 ± 0.12	0.95 ± 0.2
Bumblebee	YT Y2	0.25	1	0.62	NS	0.35 ± 0.13	0.5 ± 0.17
Hoverfly	YT Y2	0.09	1	0.77	NS	0.22 ± 0.14	0.35 ± 0.24
<i>Richness</i>							
Bee richness	PT Y1	0.01	1	0.99	NS	1.01 ± 0.12	0.94 ± 0.11
Bee richness	PT Y2	3.29	1	0.07	NS	1.19 ± 0.11	0.89 ± 0.11

Presented with mean ± standard error, chi-square X^2 , degrees freedom df, significance (NS not significant; * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$)

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s10841-022-00387-2>.

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Author contributions JGL and DG conceived the ideas and methodology; JGL collected and analysed the data, and led the writing of the manuscript. All authors contributed to revising critically to the drafts for important intellectual content and gave final approval for publication. The authors report no conflicts of interest.

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Declarations

Ethical approval This study meets all the requirements for ethical approval from the SCITEC C-REC at the University of Sussex.

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