



Pollinator-friendly management does not increase the diversity of farmland bees and wasps



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ABSTRACT

In order to reverse declines in pollinator populations, numerous agri-environment schemes have been implemented across Europe, predominantly focused on increasing the availability of floral resources. Whilst several studies have investigated how bees and wasps (aculeates) respond to management at the scale of the scheme (i.e. within the flower patch) there has been little assessment of how schemes affect diversity at the farm scale. In the current work we assessed whether farms implementing flower-rich schemes had richer aculeate communities than farms without such habitats. A total of 104 species of bee and 44 species of aculeate wasp were recorded. Farms providing flower-rich habitats had significantly greater floral abundance but there were no differences in the total number of aculeate or flowering plant species recorded compared to farms without these habitats. After accounting for differences in sample size, and contrary to expectations, farms without flower-rich habitats were significantly richer in aculeate and flowering plant species. Bumblebees (*Bombus* spp.) and honeybees (*Apis mellifera*) foraged strongly from sown flowers, but the majority of bee species preferred wild plants that are not included in flower-rich schemes such as *Heracleum sphondylium*, *Hypochaeris radicata* and *Tripleurospermum inodorum*. The creation of pollinator-friendly habitats has not increased the diversity of flowering plants and such schemes will consequently only benefit a limited suite of aculeate species. If diverse aculeate communities are to be retained and restored on farmland, agri-environment schemes that provide foraging and nesting resources for a wider range of pollinator species must be developed.

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1. Introduction

The process of agricultural intensification has resulted in simplified and less heterogeneous landscapes across Europe and North America and it is the primary driver behind long-term declines in farmland biodiversity (Benton et al., 2003; Tscharrntke et al., 2005). As with many taxa, wild bees have been negatively affected with serious declines across Europe and North America (Biesmeijer et al., 2006; Williams and Osborne, 2009; Potts et al., 2010), though there is evidence that rates of decline are slowing (Carvalho et al., 2013). These declines are of serious concern as pollinators, particularly bees, provide an important pollination service which both supports wild plant communities (Ashman et al., 2004) and affects the yield for approximately 70% of crop species worldwide, representing around 35% of total global food production (Klein et al., 2007). Up to 80% of this pollination service has been

attributed to honeybees *Apis mellifera* L. (Carreck and Williams, 1998), but more recent studies have highlighted the importance of the service provided by wild pollinators. The quality of this service depends on the diversity of the pollinator community, with richer assemblages improving yields in many crop species (Westerkamp and Gottsberger, 2000; Klein et al., 2003; Hoehn et al., 2008). Wild bee pollinators can provide the majority of crop visitation, even in contemporary intensive farming systems (Winfree et al., 2008), and enhance fruit set regardless of honeybee abundance (Garibaldi et al., 2013), leading to suggestions that the role of honeybees as agricultural pollinators has been overstated (Breeze et al., 2011; Ollerton et al., 2012). Pollinator communities are sensitive to habitat loss (Kremen et al., 2002; Steffan-Dewenter et al., 2002), and consequently the service they provide can reduce as agricultural intensification degrades the semi-natural environments upon which they depend (Garibaldi et al., 2011).

In order to reverse the decline in farmland biodiversity, agri-environment schemes are now funded across the European Union as part of the Common Agricultural Policy. These schemes offer opportunities to create pollinator-friendly habitats with the

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objectives of increasing both pollinator abundance and diversity. In England, two tiers of environmental stewardship were established in 2005. Entry Level Stewardship (ELS, [Natural England, 2013a](#)) was open to all farmers. Higher Level Stewardship (HLS, [Natural England, 2013b](#)) which was targeted to high priority areas provided greater financial rewards for more substantial and rigorous agri-environmental schemes. Most agricultural land in England is under some form of environmental stewardship with 72% of land under at least ELS management and around 21% of this total under HLS management as of 2014. ([JNCC, 2014](#)). There are three main pollinator-focused HLS options. HF4 pollen and nectar mixes are rotational plots or strips sown with a mixture of predominantly leguminous plants such as *Trifolium pratense* L. and *Trifolium hybridum* L. which tend to be resown within a 5 year period. HE10 floristically enhanced grass buffer strips are non-rotational grassland alongside fields and are composed of a mixture of grasses and plants such as *Centaurea nigra* L. and *Lotus corniculatus* L. HK6/7/8 focus on the maintenance, restoration and creation of species-rich grassland.

The response of bumblebees to the creation of flower-rich schemes has received a great deal of attention. From the initial designation of five UK bumblebees as Biodiversity Action Plan species of conservation concern, and the resulting research into their ecological requirements, it was argued that without landscape scale habitat restoration these species would continue to decline ([Edwards, 1999](#)). Many bumblebee species that have suffered substantial declines tended to collect a large proportion of pollen from Fabaceae ([Goulson et al., 2005](#)), and consequently leguminous plants became an important part of agri-environment scheme design. A number of studies have focused on the response of foraging bumblebees to these schemes and other agri-environment scheme habitats ([Edwards, 2003](#); [Pywell et al., 2006, 2011](#); [Carvell et al., 2007](#); [Holland et al., 2015](#)), but the response of other bees to these options has received much less attention and is correspondingly less well characterised, both in the UK and abroad ([Haaland et al., 2011](#)). Studies suggest that the wider bee community can respond well to management at a local level, with higher species richness recorded in targeted areas, such as field margins or meadows, compared to unmanaged control areas ([Knop et al., 2006](#); [Albrecht et al., 2007](#)). However, there has been little comparison of diversity between farms at a scale greater than that of the targeted area. It is not clear to what extent any positive benefits extend to the wider landscape, as species richness can drop off sharply outside target areas ([Albrecht et al., 2007](#); [Kohler et al., 2008](#)). If schemes are effectively increasing species diversity we would expect farms implementing appropriate management to support significantly more diverse pollinator communities. Bees and wasps can be highly aggregative, congregating on appropriate habitat ([Heard et al., 2007](#)), but not all species will be attracted to agri-environment schemes, necessitating more extensive surveying within and between targeted areas to accurately capture the community present at any one site.

In a recent survey of insect pollination scientists and conservation practitioners, the top research priority identified was to understand how important the diversity of pollinator species was to the resilience and reliability of the pollination service ([Dicks et al., 2013](#)). Other than bumblebees, there is little knowledge as to which bee and aculeate wasp species (henceforth collectively referred to as aculeates) persist on agricultural land and how they respond to agri-environment schemes. Without an appropriate evidence base, conservation interventions generally, and agri-environment schemes in particular are unlikely to be effective ([Kleijn and Sutherland, 2003](#); [Sutherland et al., 2004](#)). In this study we assess whether farms implementing pollinator-friendly management schemes support richer and more diverse aculeate

communities at the farm level. Our results have clear implications for the future management of farmland aculeate populations.

2. Materials and methods

Nine HLS and nine ELS farms were selected in Hampshire and West Sussex, UK. A map of the study area can be found in [Appendix A](#). On average, HLS farms were significantly larger (256.22 ± 37.80 ha) than ELS farms (156.67 ± 22.07 ha, t -test, $t_{16,17} = 2.565$, $p = 0.021$). The selected HLS farms had been implementing an average of 5.56 ± 0.13 ha of pollinator focused flower-rich options representing $2.17 \pm 0.05\%$ of the farm area by ownership for a minimum of three years. As the majority of farms in England are in some form of environmental stewardship, ELS management was chosen as the control group for this study. Whilst pollinator-focused flower-rich options are available as part of ELS management, the selected ELS farms were not implementing any such schemes, representing the approach of most ELS farms. Farms were predominantly arable or mixed arable/dairy with wheat, barley, oilseed rape and permanent/silage grassland as the major crops.

As the proportion of the landscape which is comprised of semi-natural habitat can affect the species richness of bees ([Steffan-Dewenter et al., 2002](#); [Holzschuh et al., 2010](#)), the farms were mapped to ensure their overall similarity. A detailed land cover map to a resolution of 2 m^2 was constructed in Google Earth (Google Inc) based on detailed surveys during the field season and satellite imagery. A 1 km buffer was drawn around the centre point of each transect covering the majority of each farm and some of the surrounding area. Semi-natural habitats fell into four broad groups (i) flower-rich agri-environment scheme grassland, (ii) flower-poor general grassland (not including permanent pasture and silage grassland, categorised as intensive grassland), (iii) hedgerows and (iv) woodland ([Table 1](#)). There was no significant difference in the proportion of the landscape covered by semi-natural habitats between farm types (GLM, $t_{16,17} = 0.147$, $p = 0.885$). The presence of water and urban structures can also affect the species richness of bees and wasps by creating moisture gradients that increase floral abundance and by providing floral resources

Table 1
Habitat composition within a 1 km radius for selected farms in Hampshire and West Sussex, UK. Means \pm 1 SE are given for nine study sites per farm type. Habitat types marked with a * were categorised as semi-natural.

Habitat type	Area (%)	Minimum (%)	Maximum (%)
<i>(a) Higher level Stewardship farms</i>			
Arable land	59.73 \pm 5.13	28.18	72.99
Flower-rich grassland*	1.77 \pm 0.41	0.38	4.36
Flower-poor general grassland*	8.36 \pm 1.43	3.02	16.19
Intensive grassland	14.15 \pm 3.34	3.57	34.32
Water	0.34 \pm 0.22	0.00	2.01
Hedgerows*	1.96 \pm 0.24	0.77	3.12
Urban	4.89 \pm 0.79	1.83	8.60
Woodland*	8.81 \pm 3.06	0.84	24.96
<i>(b) Entry level Stewardship farms</i>			
Arable land	55.76 \pm 6.06	25.08	78.23
Flower-rich grassland*	0.05 \pm 0.05	0.00	0.47
Flower-poor general grassland*	5.73 \pm 1.03	2.88	12.40
Intensive grassland	15.93 \pm 3.18	0.68	30.77
Water	1.52 \pm 1.49	0.00	13.44
Hedgerows*	2.26 \pm 0.36	0.39	3.40
Urban	4.82 \pm 0.58	2.68	7.44
Woodland*	13.92 \pm 4.41	1.52	35.17

and nesting areas (Goulson et al., 2002; Winfree et al., 2007). There was no significant difference in the proportion of the landscape covered by either of these habitat types (GLM, $t_{16,17} = 1.256$, $p = 0.232$, $t_{16,17} = 0.064$, $p = 0.949$ respectively).

A standardised 3 km transect was designed for each farm. Transects on HLS farms were designed to pass through as many pollinator-focused schemes as possible. HLS transects passed through an average of 1496 ± 148 m of flower-rich habitat in an average of 3.77 ± 0.24 discrete habitat patches. Aculeate activity was recorded along each transect following standard bee walk methodology (Carvell et al., 2007). On each 3 km transects all aculeates within 2 m of the recorder were identified to species level. Specimens which could not be immediately identified in the field were netted and later identified in the laboratory. Aculeate floral preferences were measured on the transects, with the first flowering plant species visited recorded for both observed and netted individuals. All surveys were conducted between 0930 and 1700 h when the temperature was above 13 °C with at least 60% clear sky, or above 17 °C with any level of cloud. No surveys were conducted when it was raining. On each transect, the number of flowering units of each plant species was estimated. Grasses, sedges and rushes were not recorded. This assessment followed Carvell et al. (2007) with one flower cluster (e.g. an umbel, a head, a capitulum) counted as a single unit. All aculeate and floristic surveys were conducted by the same individual (TJW) to minimise recorder bias.

Sixteen farms (8 HLS, 8 ELS) were surveyed in 2013. Transects were walked three times throughout the year, between 25th May–5th June, 26th June–15th July and 3rd–11th August. Seventeen farms (8 HLS, 9 ELS) were surveyed in 2014. Transects were walked three times throughout the year, between 17th–27th May, 21st June–9th July and 3rd–15th August.

Twelve pan traps were placed out on each farm. Pan traps consisted of 500 ml plastic bowls and were spray-painted fluorescent blue, yellow or white (Sparvar Leuchtfarbe, Spray-Color GmbH, Germany). One of each colour was attached to a wooden post of approximate height 60 cm. On HLS farms two such posts were placed in a flower-rich margin and two were placed in a flower-poor general grass margin. On ELS farms two posts were placed in each of two separate flower-poor general grass margins. Posts in the same margin were separated by a distance of 25 m. Sampled margins were no closer than 200 m from each other. Traps were filled with approximately 400 ml of water with a few drops of unscented washing up liquid (Surcare Sensitive, UK). Traps were left out for 96 h before being collected.

Sixteen farms (8 HLS, 8 ELS) were surveyed in 2013. Traps were set three times between 10th–14th June, 8th–12th July and 1st–5th of August. Fourteen farms (7 HLS, 7 ELS; all of these farms were surveyed with transect walks) were surveyed in 2014. Trap placement was staggered over two days, with half of the farms (4 HLS, 3 ELS) trapped on day one and the other half of the farms (3 HLS, 4 ELS) trapped on day two. Traps were set three times between 29th/30th May–2nd/3rd June, 10th/11th–14th/15th July and 12th/13th–16th/17th August. Invertebrate samples were stored in 70% ethanol and pinned prior to identification.

Differences in aculeate and floristic species richness between farm types were investigated using rarefaction and extrapolation curves. Both the transect and the pan trap techniques produce replicated sample-based abundance data, where it is the sampling unit rather than the individual species that is sampled randomly and independently (Gotelli and Colwell, 2001). Following Colwell et al. (2012), this sample-based abundance data was converted to sample-based incidence data to calculate incidence-based extrapolation curves using a Bernoulli product model. Differences in the number of individuals per sampling unit is accounted for by rescaling the X-axis to individuals (Gotelli and Colwell, 2001), allowing

for a more meaningful comparison between assemblages of differing sample sizes. Rarefaction curves with 95% confidence intervals were produced using EstimateS 9.1.0 (Colwell, 2013). Curves were calculated without replacement and were extrapolated to twice the number of collected samples. Similarity in community composition between different farm types was assessed using Simpson's similarity index (Simpson 1960). This index was chosen as it accounts for variable sampling effort between sites, as not every site was surveyed with transects or pan traps in every year. Community similarity within and between farm management types was compared using a permutation ANOVA following Chase (2007).

The impact of the proportion of the landscape represented by semi-natural habitats (flower-rich grassland, flower-poor grassland, hedgerow, woodland and all habitats combined) on the number of aculeate species recorded on the transect and in the pan traps was investigated using Generalised Linear Models. No significant relationships were found (Appendix B).

Differences in total number of aculeate and flowering plant species between farm types were investigated using Generalised Linear Mixed-Effect Models (GLMMs). The total number of species recorded by each sampling technique was calculated for each farm in each year. Sampling year was included in the model as a random factor to account for temporal pseudoreplication. Differences in floral abundance between management types was tested with a GLMM with sampling round nested within sampling year included as random factors to account for the temporal data structure. The impact of floristic richness on aculeate richness was also tested using a GLMM. Sampling year was included in the model as a random factor. Models were fitted with a Gaussian error distribution unless found to be significantly non-normal. Where models were significantly non-normal they were fitted with Poisson error distributions though in all cases they were found to be significantly over dispersed and so were fitted with negative binomial error distributions instead. Final models were compared by ANOVA with a null model containing the same random factors to test for significance. Differences in the proportion of visits to plants sown as part of HLS management by different bee groups were tested using 2-sample tests for equality of proportions. All data analysis was conducted in R version 3.1.1 (R Development Core Team) using the package fossil to calculate Simpson's similarity indices, the package coin for the permutation ANOVA and the package lme4 for the GLMMs.

3. Results

A total of 16,821 aculeates of 148 species were recorded comprising 104 species of bee and 44 species of wasp, including 21 species of conservation concern (Table 2). This follows the rarity classifications in Falk (1991) but excludes *Lasioglossum malachurum* Kirby, *L. pauxillum* Schenck and *Bombus rupestris* Fabricius which have become much more widespread since this classification and no longer merit nationally scarce status (S. Falk, pers. comm.). The bumblebee *B. ruderarius* Müller is also included due its recent decline in the UK and designation as a Biodiversity Action Plan species. Both *Nomada flava* Panzer and *N. panzeri* Lepelletier were recorded but as the males are indistinguishable these two species were merged for analysis purposes.

On the transects 12,136 aculeates of 112 species were recorded, with a total of 82 and 89 species found on HLS and ELS farms, respectively (Appendix C). There was no difference in the total number of species recorded between farm types ($\chi^2 = 0.1$, $p = 0.767$). Rarefaction analysis showed that ELS farms were significantly richer than HLS farms after approximately 2000 sampled individuals (Fig. 1). The pan traps caught 4,685 aculeates of 115 species, with a total of 89 and 98 species found on HLS and

Table 2

Species of conservation concern recorded during the survey and the number of individuals recorded on HLS and ELS farms. Rarity status from Falk (1991), also includes Biodiversity Action Plan (BAP) species. RDB1 = Red Data Book 1 species in danger of extinction, RDB3 = Red Data Book 3 species estimated to occur in fewer than 15 × 10 km squares, Na = Nationally Scarce A species estimated to occur within 16–30 × 10 km squares and Nb = Nationally Scarce B species estimated to occur within 31–100 × 10 km squares.

Species	Rarity	Number of individuals recorded on HLS farms	Number of individuals recorded on ELS farms
<i>Andrena alfenella</i> (Perkins 1914)	RDB3	2	1
<i>Andrena florea</i> (Fabricius 1793)	RDB3	1	7
<i>Andrena fulvago</i> (Christ 1791)	Na		1
<i>Andrena humilis</i> (Imhoff 1832)	Nb		2
<i>Andrena labiata</i> (Fabricius 1781)	Na		1
<i>Andrena minutuloides</i> (Perkins 1914)	Na	1	
<i>Bombus ruderarius</i> (Müller 1776)	BAP	4	
<i>Bombus ruderatus</i> (Fabricius 1775)	Nb/BAP	5	
<i>Hoplitis claviventris</i> (Thomson 1872)	Nb	1	
<i>Hylaeus cornutus</i> (Curtis 1831)	Na	15	
<i>Hylaeus signatus</i> (Panzer 1798)	Nb		1
<i>Lasioglossum puncticolle</i> (Morawitz 1872)	Nb	4	
<i>Lasioglossum xanthopus</i> (Kirby 1802)	Nb	1	
<i>Melitta tricincta</i> (Kirby 1802)	Nb	18	43
<i>Microdynerus exilis</i> (Herrich-Shäffer 1839)	Nb	1	1
<i>Mimumesa unicolor</i> (Vander Linden 1829)	Na		2
<i>Nomada flavopicta</i> (Kirby 1802)	Nb		1
<i>Nomada guttulata</i> (Schenck 1861)	RDB1	1	
<i>Nyssus trimaculatus</i> (Rossi 1790)	Na	2	
<i>Osmia bicolor</i> (Schrank 1781)	Nb	2	1
<i>Sphecodes niger</i> (von Hagens 1874)	RDB3		3

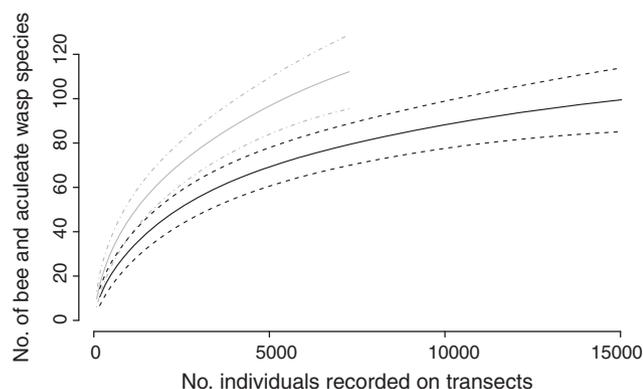


Fig. 1. Rarefaction and extrapolation curves for aculeate richness recorded on the transects for ELS farms (upper grey line with dashed and dotted-line 95% confidence intervals) and HLS farms (lower black line with dashed-line 95% confidence intervals).

ELS farms, respectively (Appendix C). There was no difference in the total number of species recorded between farm types ($\chi^2 = 1.6$, $p = 0.203$). Rarefaction analysis showed that there was no significant difference in aculeate richness between farm types, though ELS farms tended to have more species (Fig. 2). Aculeate community composition of farms did not more closely resemble farms of the same management type than farms of the other management type (permutation ANOVA, $Z = 1.594$, $p = 0.108$).

On the transects 9.69 million flowering units of 237 species of flowering plant were recorded, with a total of 198 and 190 flowering plant species found on HLS and ELS farms respectively (Appendix D). Floral abundance was significantly greater on HLS farms than ELS farms ($502,523 \pm 74,324$ flowering units/year

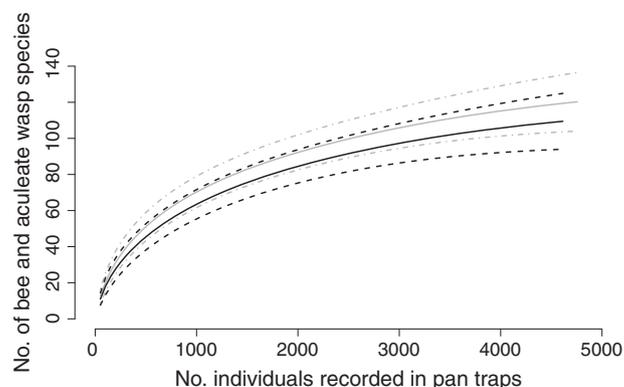


Fig. 2. Rarefaction and extrapolation curves for aculeate richness recorded in the pan traps for ELS farms (upper grey line with dashed and dotted-line 95% confidence intervals) and HLS farms (lower black line with dashed-line 95% confidence intervals).

against $93,608 \pm 22,703$ flowering units/year, $\chi^2 = 49.0$, $p < 0.001$). There was no difference in the number of flowering plant species between farm types ($\chi^2 = 1.1$, $p = 0.293$), but rarefaction analysis showed that ELS farms were significantly richer in flowering plant species (Fig. 3). Floristic richness had a significant effect on the richness of bees recorded on the transects, but not the richness of aculeate wasps ($\chi^2 = 12.6$, $p < 0.001$, $\chi^2 = 0.6$, $p = 0.797$, respectively). Floristic richness had no effect on the richness of bees recorded in the pan traps and a significantly negative effect on the richness of aculeate wasps ($\chi^2 = 0.0$, $p = 0.948$, $\chi^2 = 6.6$, $p = 0.010$, respectively).

On the transects 11,661 foraging trips to 124 species of flowering plant were recorded by 81 species of bee. Plants sown as part of HLS management were visited extensively by both bumblebees (*Bombus* spp.) and honeybees (*A. mellifera*), with visits to these plants accounting for 69% ($n = 6441/9288$) and 68% ($n = 1147/1698$) of total visits, respectively, a similar overall proportion ($\chi^2 = 2.1$, $p = 0.149$), in particular *C. nigra* L. ($n = 4558$), *T. hybridum* L. ($n = 704$) and *L. corniculatus* L. ($n = 523$). However, only 32% ($n = 23/72$) of the non-corbiculate bee species recorded on the transects were ever observed to visit sown flowers (including visits to the sown species growing wild on both farm types), these plants representing 33% ($n = 224/675$) of total visits, a significantly lower proportion than for bumblebees ($\chi^2 = 370.0$, $p < 0.001$) and honeybees ($\chi^2 = 232$, $p < 0.001$). The sweat bee *L. malachurum* accounted for 72% ($n = 157/218$) of these visits to sown flowers. Only six species of non-corbiculate bees were observed visiting Fabaceae flowers (a dominant component of most sown flower mixes), representing 3% ($n = 23/675$) of total visits. Excluding sown species,

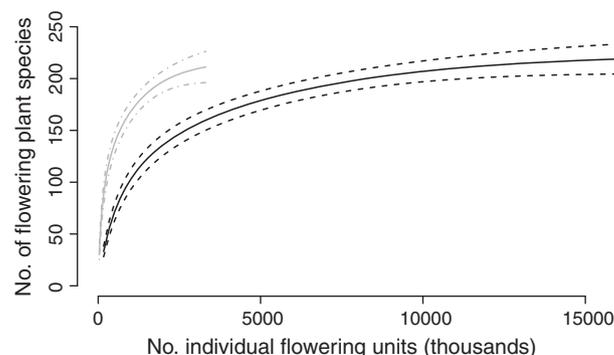


Fig. 3. Rarefaction and extrapolation curves for floristic richness recorded on the transects for ELS farms (upper grey line with dashed and dotted-line 95% confidence intervals) and HLS farms (lower black line with dashed-line 95% confidence intervals).

Table 3

The ten most popular forage plants for different aculeate groups. Plants sown as part of pollinator friendly management are highlighted in bold.

<i>Bombus</i> spp.	Visits (%)	<i>Apis mellifera</i>	Visits (%)	Other bees	Visits (%)	Wasps	Visits (%)
<i>Centaurea nigra</i>	43.21	<i>Centaurea nigra</i>	32.08	<i>Centaurea nigra</i>	18.52	<i>Heracleum sphondylium</i>	48.35
<i>Lotus corniculatus</i>	5.23	<i>Trifolium hybridum</i>	15.60	<i>Heracleum sphondylium</i>	10.22	<i>Daucus carota</i>	18.68
<i>Trifolium pratense</i>	5.23	<i>Heracleum sphondylium</i>	8.83	<i>Hypochaeris radicata</i>	8.00	<i>Angelica sylvestris</i>	10.98
<i>Trifolium hybridum</i>	4.73	<i>Melilotus officinalis</i>	6.24	<i>Leucanthemum vulgare</i>	7.11	<i>Cirsium arvense</i>	6.59
<i>Cirsium vulgare</i>	3.62	<i>Rubus fruticosus</i> agg.	5.06	<i>Tripleurospermum inodorum</i>	6.52	<i>Pastinaca sylvestris</i>	4.40
<i>Stachys sylvatica</i>	3.47	<i>Phacelia tanacetifolia</i>	3.65	<i>Crepis capillaris</i>	6.07	<i>Achillea millefolium</i>	2.20
<i>Arctium minus</i>	2.99	<i>Sonchus arvensis</i>	3.59	<i>Odontites verna</i>	4.89	<i>Scrophularia auriculata</i>	2.20
<i>Lamium album</i>	2.84	<i>Origanum vulgare</i>	2.77	<i>Pulcharia dysenterica</i>	4.59	<i>Anthriscus sylvestris</i>	1.10
<i>Dipsacus fullonum</i>	2.80	<i>Pulcharia dysenterica</i>	2.77	<i>Picris hieracioides</i>	3.26	<i>Euphorbia amygdaloides</i>	1.10
<i>Trifolium repens</i>	2.77	<i>Reseda lutea</i>	2.65	<i>Taraxacum</i> agg.	3.26	<i>Leucanthemum vulgare</i>	1.10

non-corbiculate bees foraged heavily from Asteraceae and Apiaceae, accounting for 73% ($n = 261/457$ and $n = 74/457$) of total visits, in particular *Heracleum sphondylium* L. ($n = 69$), *Hypochaeris radicata* L. ($n = 54$) and *Tripleurospermum inodorum* (L.) Schultz Bip ($n = 44$) (Table 3).

Wasps rarely visit flowers, instead provisioning their offspring with invertebrates. A total of 91 visits to 13 species of flowering plant were recorded by 17 species of wasp. Plants sown as part of HLS management were seldom visited by wasps, accounting for 22% ($n = 20/91$) of total visits. Wasps predominantly visited Apiaceae, accounting for 85% ($n = 77/91$) of total visits, in particular *H. sphondylium* ($n = 44$), *Daucus carota* L. ($n = 17$) and *Angelica sylvestris* L. ($n = 10$).

4. Discussion

The 104 bee species recorded in the current work represent approximately half of the bee species found in the south-east of England, with 198 and 199 species recorded since 1970 from the neighbouring counties of Surrey and Kent, respectively (Baldock, 2008). Given that many habitat specialists are unlikely to be found on farmland this number is a substantial fraction of all possible species, with many European studies only recording around 40–70 bee species (e.g. Holzschuh et al., 2008; Kovács-Hostyánszki et al., 2013; Le Féon et al., 2013), despite surveying a more extensive European fauna, though see Batáry et al. (2010). Direct surveys for farmland wasps are uncommon, with most studies assessing trap nesting wasps only (e.g. Holzschuh et al., 2010). Bee faunas are known to be locally diverse, variable in space and time and rich in rare species with up to 50% of species represented by singletons, even in studies recording several thousand individuals (Williams et al., 2001). Our results show that reasonably high aculeate diversity, including species of national conservation concern, can persist in agricultural environments containing 21–22% semi-natural habitat. This proportion of semi-natural habitat is high, as areas where comparable studies have been carried out in continental Europe often have considerably less semi-natural habitat, typically 5–10% (e.g. Steffan-Dewenter et al., 2002; Westphal et al., 2003; Herrmann et al., 2007). It has been argued that landscape complexity is more important than local management, with positive benefits of management seen only in simple landscapes (Tscharntke et al., 2005). The addition of 5.5 ha of flower-rich habitat to a landscape which already has around 20% semi-natural habitat may not be enough to make a significant difference to existing aculeate populations. It may be the case that current HLS prescriptions will significantly increase floristic and aculeate diversity in simpler, more intensively farmed landscapes comprising around 5–10% semi-natural habitat, but this remains to be tested.

The loss of meadows and leys rich in Fabaceae has been particularly associated with bumblebee declines (Carvell et al., 2006; Williams and Osborne, 2009), and many species that have declined substantially have likewise been shown to collect a larger

proportion of their pollen from Fabaceae (Goulson et al., 2005; Kleijn and Raemakers, 2008). It has also been argued that the abundance of the most rewarding forage plants is more important than the overall diversity of flowering plants for maintaining diverse bumblebee communities (Williams, 1989). As a result, increasing floral abundance, particularly the abundance of leguminous forage, and observing the response of foraging bumblebees has been an important part of the design and assessment of pollinator-friendly agri-environment schemes (Edwards, 2003; Pywell et al., 2006; Carvell et al., 2007). However, there has been relatively little assessment of the response of other wild bees to these options (Haaland et al., 2011), despite the fact that these other bees make up the large majority of species diversity. Where studies have been conducted, non-corbiculate bees show little preference for sown species. Carreck and Williams (2002) recorded only six species of non-corbiculate bees visiting annual wildflower mixes, representing just 1% ($n = 223/21,841$) of total visits by all bees. Our results demonstrate that whilst bumblebees and honeybees foraged strongly from sown flowers, the majority of non-corbiculate bee species were not observed visiting these plants, even when growing wild as part of an existing flora. Non-corbiculate species predominantly favoured Asteraceae and Apiaceae growing in wayside and hedgerow habitats. Whilst wasps visit flowers much less frequently, similar patterns were observed, with the majority of visits being to non-scheme plants, particularly Apiaceae.

Bee community richness has been frequently linked to floristic richness (Tscharntke et al., 1998; Potts et al., 2003; Batáry et al., 2010), and a loss of plant species has been associated with a decline in bee diversity in agricultural habitats (Banaszak, 1992; Biesmeijer et al., 2006; Scheper et al., 2014). Options for establishing floristically enhanced habitats as part of HLS management typically involve sowing mixes containing around 20 native species of wildflowers (Carvell et al., 2007; Pywell et al., 2011). However, the systematic addition of inorganic fertilisers over the past 60 years has negatively affected floral diversity in field margins and other non-cropped areas, primarily by increasing the competitiveness of coarse grasses (Kleijn and Verbeek, 2000). High nutrient levels are not easily reduced and grasslands can suffer chronically low levels of biodiversity for decades after enrichment has ceased (Isbell et al., 2013). Consequently, establishing diverse floral communities on enriched soils faces a number of difficulties, with the most successful programs requiring deep cultivation and sowing of a selection of ecologically appropriate species (Pywell et al., 2002).

In addition to foraging resources, aculeates require suitable nesting habitat to complete their lifecycles. Cavity nesting bees and wasps are generally thought to be limited by cavity availability (Holzschuh et al., 2010), and the provision of trap nests can significantly increase their abundance (Gathmann and Tscharntke, 1997). Whilst the majority of bees are ground nesters, the availability of nesting resources can play a small but important role in organising bee communities (Potts et al., 2005). There has been

little assessment of how well agri-environment schemes can provide appropriate habitat for nesting bees other than for bumblebees (Lye et al., 2009). Deliberate habitat manipulation to create nesting habitat for aculeates on agricultural land has been advocated (Kremen et al., 2007), but there are currently no specific schemes under HLS management with this aim in mind.

Many farmers lack the requisite knowledge to restore grassland communities and the created habitats are often dominated by relatively few species, often *C. nigra*, *L. corniculatus* and *Trifolium* spp. Whilst the creation of flower-rich habitats significantly increased floral abundance, in our study it did not significantly increase floral diversity at the farm level. Whilst the flower species that are currently sown and establish well in flower-rich margins provide attractive forage for bumblebees and honeybees, they are of limited use to the majority of bee species. Given that current schemes were designed around the foraging requirements of bumblebees, whilst they can significantly increase the population size of common bumblebee species (Wood et al., 2015), it is perhaps not surprising that they are only visited by a restricted suite of aculeates. Increasing the number of pollinators is important, as larger wild bee populations can significantly increase the quality of the pollination service provided (Blaauw and Isaacs, 2014). Recent work has also highlighted the importance of a diverse pollinator community for providing a high quality pollination service (Westerkamp and Gottsberger, 2000; Klein et al., 2003; Hoehn et al., 2008; Garibaldi et al., 2013). The extent to which agri-environment schemes should focus on ecosystem service delivery or biodiversity conservation is not clear, with most schemes benefiting generalist pollinators that can respond rapidly to habitat creation (Scheper et al., 2013). Since both pollinator diversity and abundance are important for crop pollination (Garibaldi et al., 2014), such objectives may not be greatly opposed. Bumblebees and honeybees are important pollinators, and increasing the size of their populations is an appropriate goal, but a better appreciation that current schemes predominantly benefit corbiculate bees and are of limited use to the wider aculeate community is also desirable. Complementary 'non-corbiculate-focused' schemes should be developed and implemented, as in their current form pollinator-focused agri-environment schemes do not provide suitable resources to support a significantly more diverse aculeate community.

5. Conclusions

Increasing the size of pollinator populations is an important aim of environmental stewardship, but management that focuses purely on increasing floral abundance may fail to support a diverse aculeate community. Despite the lack of evidence that Fabaceae-rich options provide resources for non-corbiculate bees, they are the most highly appraised pollinator-focused agri-environment scheme (Breeze et al., 2014). Our results show that current management techniques described as pollinator-friendly appear to benefit only a limited suite of aculeate species, predominantly a small number of bumblebee species and honeybees. The majority of bee species foraged heavily from naturally regenerating Asteraceae and Apiaceae in non-scheme areas; it is these and other non-Fabaceae plant groups that also need to be encouraged to maintain a diverse bee community. Consequently, agri-environment schemes that enable land managers to create and maintain local floral diversity should be developed and investigated to more effectively retain and restore aculeate diversity on farmland.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.biocon.2015.04.022>.

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