Appendix 4

Received: 9 August 2021 | Accepted: 19 January 2022
DOI: 10.1111/1365-2664.14132

Journal of Applied Ecology BRITISH ECOLOGICAL SOCIETY

RESEARCH ARTICLE

The disproportionate value of 'weeds' to pollinators and biodiversity

Nicholas J. Balfour | Francis L. W. Ratnieks

Laboratory of Apiculture & Social Insects, School of Life Sciences, University of Sussex, Brighton, UK

Correspondence
Nicholas J. Balfour
Email: n.balfour@sussex.ac.uk

Funding information
Rowse Honey Ltd.

Handling Editor: Benjamin Woodcock

Abstract

1. Agricultural intensification has been implicated in global biodiversity declines. In the European Union, agri-environmental schemes are designed to address this. For pollinating insects, funding has been provided to sow wildflower mixes. However, previous research indicates that a suite of agricultural weeds are also of great importance to pollinators.
2. Here, we compare the biodiversity associated with the species which are considered harmful to agricultural production and legally deemed as 'injurious' by the United Kingdom 1959 Weeds Act (common ragwort *Jacobaea vulgaris*, creeping thistle *Cirsium arvense*, spear thistle *C. vulgare*, curled dock *Rumex crispus* and broadleaved dock *R. obtusifolius*), with plant species recommended for pollinator-targeted agri-environmental options.
3. In our field study, the abundance and diversity of pollinators visiting the weed species averaged twice that of the recommended plants and included the main insect orders (Coleoptera, Diptera, Hymenoptera and Lepidoptera). This relationship was also seen in a meta-analysis of literature data, which indicates that fourfold more flower-visitor species and fivefold more conservation-listed species are associated with the weeds. Additionally, the literature shows that twice the number of herbivorous insect species are associated with these plants.
4. We suggest that several factors are responsible for this pattern. Injurious weed species are widely distributed, their flower morphology allows access to a wide variety of pollinator species, and they produce, on average, four times more nectar sugar than the recommended plant species.
5. Freedom of information requests to public bodies such as local councils, Natural England and Highways England indicate that *c.* £10 million per year is spent controlling injurious weeds. Meanwhile, the cost of the four pollinator-targeted agri-environmental options in the United Kingdom exceeds £40 m annually.
6. *Synthesis and applications*. Our results clearly show that weeds have an under-appreciated value to biodiversity. Unfortunately, current UK agricultural policy encourages neither land sparing for nor land sharing with weeds. The UK government is, however, currently committed to overhauling agricultural payments to

encourage more wildlife- and climate-friendly practices. Thus, the challenge of reconciling the conflicts between agricultural production and these native and biodiverse species should be a renewed priority to land managers, researchers and policymakers.

KEYWORDS
agriculture, biodiversity, conservation, ELMS, herbivorous insects, pollinators, weeds, wildflower mixes

1 | INTRODUCTION

The importance of animal pollinators to agricultural productivity and as a component of biodiversity is well-understood (Klein et al., 2007; Ollerton et al., 2011). However, declines in the abundance and distribution of many pollinator species have prompted concerns for the maintenance of their pollination services (e.g. Goulson et al., 2015; Bartomeus et al., 2019). While several causes have been identified, there is a general consensus that agricultural intensification is a key factor (Ollerton et al., 2014; Goulson et al., 2015; Balfour et al., 2018).

The Rio Convention on the Conservation of Biodiversity and the Aichi targets were agreed upon by most countries, including the United Kingdom, in 2010. The United Kingdom's commitment to these goals and targets are set out in the Strategic Plan for Biodiversity 2011–2020, which has recently been updated (DEFRA, 2020a). In countries with fragmented natural habitats, and where most of the land area is used for agriculture, such as the United Kingdom (75%; DEFRA, 2020b), there is a need to develop practical means to maintain biodiversity within the farmed landscape (e.g. Kremen and Merenlender, 2018; Balfour et al., 2021).

In the European Union and the United Kingdom, financial support has been provided through agri-environmental schemes to encourage farmers to cater for wildlife in their land management. For pollinating insects, funding has been available to create or maintain habitats and nectar/pollen sources by sowing wildflower seed mixes in dedicated areas, or strips within arable land. Several studies have indicated that these seeded areas are beneficial to a suite of, usually common, pollinators (e.g. Haaland et al., 2011; Scheper et al., 2013; Wood et al., 2017; Sutter et al., 2018; Gasner et al., 2021). The cost of the four pollinator-targeted agri-environmental options in the United Kingdom (AB1, Nectar flower mix; AB8, Flower-rich margins and plots; AB16: Autumn sown bumblebird mix; and GS4, Legume and herb-rich swards; DEFRA, 2020c) is estimated to exceed £40 m annually (Rayment, 2017).

Weeds, by contrast, need no encouragement in the agricultural landscape. The UK Weeds Act (UKGOV, 1959) classes five common and widespread native species (*Jacobaea vulgaris*, *Cirsium arvense*, *C. vulgare*, *Rumex crispus* and *R. obtusifolius*) as 'harmful to agricultural production or [potentially] harmful if eaten by animals' (DEFRA, 2003) and aims to halt their spread. The act can require landowners to take measures to prevent their spread and potentially carries a £1,000 fine if they fail to act. Moreover, the UK Ragwort Control Act (UKGOV, 2003a) established a code of practice to control ragwort, *J. vulgaris* [Correction added on 4 April 2022, after first online publication: The word "requires" has been replaced to "can require", "can carry" to "potentially carries" in the sentence.]. However, these weed species support a great diversity of wildlife, including insects, which use them as larval food plants (e.g. Bacon et al., 2003). Furthermore, there is a growing body of evidence which indicates that a suite of common agricultural weeds are also of great importance to flower-visiting insects (Feber et al., 1996; Steffan-Dewenter and Tscharntke, 2001; Kremen et al., 2002; Marshall et al., 2003; Pywell et al., 2005; Carvell, et al., 2006; Carvalheiro et al., 2011; Pocock et al., 2012; Rollin et al. 2013; Balfour et al., 2015; Requier, et al., 2015; Redhead et al., 2018; Timberlake et al., 2018) and herbivorous insects (reviewed in Norris and Kogan, 2005; Barberi et al., 2010).

Here, we use multiple datasets to evaluate the biodiversity value of the injurious weeds versus other British wildflower species. First, we used data from our field study to compare the abundance and diversity of flower-visitors to the three insect-pollinated injurious weeds (*J. vulgaris*, *C. arvense* and *C. vulgare*) versus flower species recommended for pollinator-targeted agri-environmental options. Second, a meta-analysis of literature data from two databases was used to evaluate the diversity of both flower-visiting and herbivorous species associated with British plants. Lastly, Freedom of Information requests were made to British public bodies to estimate the cost of controlling the injurious weeds.

2 | MATERIALS AND METHODS

2.1 | Field study: flower-visitor abundance and diversity

To compare the abundance, diversity and species assemblage evenness (Shannon–Wiener diversity index) of insects visiting the plant species recommended for pollinator-targeted agri-environmental options (recommended plants; Countryside Stewardship grant options: AB1, Nectar flower mix; AB8, Flower-rich margins and plots; AB16: Autumn sown bumblebird mix; and GS4, Legume and herb-rich swards; DEFRA, 2020c) to that of the three insect-pollinated injurious weeds (injurious weeds; *J. vulgaris*, *C. arvense* and *C. vulgare*), we observed patches of flowers during 12–15 censuses, per species. We counted all insects visiting each of the study plant

TABLE 1 The 41 study plant species, their designation (either classed as injurious weeds in the 1959 Weeds Act or recommended for pollinator-targeted agri-environmental options) and their distribution (number of UK hectads, i.e. 10 × 10 squares). From the field survey (Field): each plant's mean flower-visitor (FV) abundance and mean FV diversity (Shannon–Wiener diversity index). From the Database of Pollinator Interactions (DoPI): the number of FV species observed and estimated, FV diversity (Shannon Wiener) and number of conservation-listed FV species per plant species. From the Database of Insects and their Food Plants (DBIF): the number of herbivorous species and conservation-listed herbivorous species associated with each plant species

Plant species	Designation	Distribution	Field FV Abundance	Field FV Shannon	DoPI Observed	DoPI Estimated	DoPI Shannon	DOPI Conservation Species	DBIF Herb Insects	DBIF Conservation Species
Achillea millefolium	AB1, AB8	2,774	4.0	1.0	69	156.7	3.8	4	71	14
Agrimonia eupatoria	AB8	1,859	0.9	0.2	8	13.7	2.0	0	9	0
Anthyllis vulneraria	AB8	1,798	NA	NA	6	10.2	1.6	2	17	6
Centaurea nigra	AB1, AB8	2,658	5.6	1.1	121	196.6	2.8	10	43	5
Centaurea scabiosa	AB8	1,239	2.7	0.7	15	34.2	2.6	2	22	5
Cichorium intybus	GS4	1,312	NA	NA	39	60.3	1.5	0	5	1
Cirsium arvense	Injurious Weed	2,736	7.9	1.1	197	397.0	3.8	5	31	2
Cirsium vulgare	Injurious Weed	2,789	4.5	1.0	84	240.3	3.2	2	27	3
Conopodium majus	AB8	2,520	NA	NA	17	48.0	2.8	2	3	0
Daucus carota	AB1, AB8	1,845	6.2	1.2	87	168.2	3.2	9	27	4
Galium mollugo	AB8	1,665	1.9	0.4	21	211.9	3.0	0	42	3
Galium verum	AB8	2,516	0.7	0.1	15	49.6	2.3	0	50	5
Geranium pratense	AB8	1,383	NA	NA	9	12.8	2.0	2	4	2
Knautia arvensis	AB8	1,707	2.8	0.8	57	95.0	3.6	1	15	4
Lathyrus pratensis	AB8	2,636	NA	NA	10	22.4	0.8	1	22	2
Leontodon hispidus	AB8	1,702	NA	NA	19	27.0	1.7	1	11	1
Leucanthemum vulgare	AB1, AB8	2,532	1.7	0.3	184	340.1	3.6	5	40	4
Lotus corniculatus	AB1,AB8, AB16, GS4	2,801	2.0	0.3	38	63.8	0.9	4	60	17
Malva moschata	AB1, AB8	1,423	4.3	0.7	12	58.7	2.4	0	1	0
Medicago lupulina	AB1	2,064	NA	NA	21	75.9	2.7	1	21	2
Medicago sativa	AB1, AB16	1,065	NA	NA	12	72.5	2.5	0	18	2
Melilotus officinalis	AB1	1,142	NA	NA	NA	NA	NA	0	14	1
Origanum vulgare	AB8	1,148	7.8	1.3	16	109.1	2.6	0	25	5
Phacelia tanacetifolia	AB16	0	NA	NA	NA	NA	NA	0	1	0
Plantago lanceolata	AB8	2,804	NA	NA	NA	NA	NA	NA	39	4
Primula veris	AB8	1,632	NA	NA	5	10.3	1.3	1	10	2
Prunella vulgaris	AB8	2,783	NA	NA	15	39.4	2.2	2	24	3
Ranunculus acris	AB8	2,780	NA	NA	125	183.5	3.0	5	17	1
Rhinanthus minor	AB8	2,629	NA	NA	4	6.5	1.2	0	9	0
Rumex acetosa	AB8	2,790	NA	NA	NA	NA	NA	NA	27	3
Rumex crispus	Injurious Weed	2,724	NA	NA	NA	NA	NA	NA	10	1
Rumex obtusifolius	Injurious Weed	2,746	NA	NA	NA	NA	NA	NA	69	7
Sanguisorba minor	AB8	1,216	NA	NA	3	5	1.1	0	14	3
Sanguisorba officinalis	AB8	1,246	NA	NA	NA	NA	NA	0	7	0
Jacobaea vulgaris	Injurious Weed	2,725	6.1	1.4	152	377.4	4.2	18	60	8
Thymus serpyllum	AB16	9	NA	NA	NA	NA	NA	0	6	1
Trifolium hybridum	AB1, AB16	1,940	NA	NA	1	1	0	0	2	0
Trifolium incarnatum	AB16	196	NA	NA	NA	NA	NA	0	7	0
Trifolium pratense	AB1, AB8, AB16, GS4	2,745	1.2	0.2	43	73.0	2.5	5	35	4
Vicia cracca	AB8	2,647	NA	NA	15	20.9	1.5	0	18	2
Vicia sativa	AB1, AB16	1,930	1.8	0.2	17	117.2	2.7	0	2	0

species (Table 1) during weather conditions suitable for all flower-visitor activity (≥16°C and light wind). Data were collected between 10:00 and 16:00 hr during 15 days, from 7 July to 18 August 2020 at six sites in Brighton, East Sussex, United Kingdom: (a) Woodingdean Park (50.8408, −0.0687), (b) University of Sussex Campus (50.8712, −0.0929), (c) Stanmer Park Local Nature Reserve (50.8663, −0.0959), (d) Castle Hill National Nature Reserve (50.8494, −0.0576), (e) University of Sussex Field Trials Plot (50.8718, −0.0841) and (f) Sheepcote Valley Local Nature Reserve (50.8285, −0.0964). All sites were predominated by unimproved calcareous grassland with varying degrees of woodland or shrub and self-sown flowers. Three of these were former pasture fields, now used as amenity grassland (a–c) and three are pastoral fields (c–f).

Patches of the study species were located by walking from the approximate centre of each study site at a randomly generated compass angle. Patches ≤5 m to either side of the observer, in full flower, and with ≥25% cover within a 1 m^2 quadrat were studied. First, a 1 m^2 quadrat was placed over the part of the patch with the greatest number of open flowers. We then estimated: (a) the percentage cover of the study plant species within the quadrat, using 5% intervals; and (b) the area (m^2) of the study plant within a 3 m radius, using 0.5 m^2 intervals. All flower-visitors observed actively foraging on the patch were then recorded for 5 min. Insects were identified to species or assigned a morphospecies name. Care was taken not to record the same individual more than once per sampling occasion. Where necessary, this was facilitated by noting unique individual features (e.g. pollen on the head or stored in corbiculae, wing damage, body size etc.). When needed for identification, individuals (n = 35) were collected for microscopic examination, using the dichotomous keys of Unwin (1984), Falk (2015), and Ball and Morris (2015). Each plant species was studied once or twice per site visit. The minimum distance between two consecutively studied patches was 10 m.

To determine the percentage of land cover types within a 1 km radius of the study sites, we used the data and land cover categories from the 2019 Land Cover Map (Rowland et al., 2017). Land cover (Appendix S1) and the distance of the study patch to the nearest study site boundary were mapped with QGIS version 2.20.3 (QGIS Development Team, 2021).

2.2 | Literature and database analyses

To assess whether the flower-visitor data generated in our field study were consistent with that in the scientific literature, we used the datasets available from the Database of Pollinator Interactions (DoPI, https://www.dopi.org.uk/; Balfour et al., in prep). DoPI is built from a systematic review of the scientific literature and unpublished datasets. It contains records detailing over 320,000 interactions between British plant and flower-visitor species (or genera), together with associated metadata. These include records from >300 datasets, comprising 1,888 pollinator species and 1,241 plant species, totalling >17,000 pairwise species interactions. Prior to analysis, the database was filtered to only include studies in which all plants (i.e. transects) and pollinators were recorded. An abundance-based species diversity estimator (Chao1 Estimator; Hsieh et al., 2016) was used to estimate asymptotic species richness of the three insect-pollinated injurious weeds and the native insect-pollinated recommended plant species. This method uses the number of rare taxa, singletons and doubletons, to estimate the number of missing species. To compare the number of British herbivorous insect species associated with the native recommended plants to the five injurious weeds, we used data from the Database of Insects and their Food Plants (DBIF, https://www.brc.ac.uk/dbif/; Padovani et al., 2020).

To calculate the number of insect species with conservation designations per plant species, we compiled a list of IUCN GB Critically Endangered, Endangered, Vulnerable and/or Nationally Rare/Scarce: Coleoptera (Hyman, 1992, 1994), Diptera (Falk, 1991a; Ball and Morris, 2014), Hymenoptera (Falk, 1991b) and Lepidoptera (Hadley, 1984; Parsons, 1984). These data were used to generate a list of interactions from all data stored in DoPI and DBIF. Geographical distributions of the study plant species were used in the analysis of both databases. This was achieved by extracting the number of unique hectads (10 km × 10 km square) per plant species from PLANTATT (Hill et al., 2004).

2.3 | Auxiliary information: financial costs of controlling injurious weeds and nectar production

To estimate the annual financial costs to public bodies of controlling the injurious weeds, we made Freedom of Information requests to: (a) 55 randomly selected Local, Regional and Unitarian councils in England, Scotland, Wales and Northern Ireland, (b) Network Rail, (c) Highways England, Transport Scotland, Transport for Wales and Northern Ireland Department of Infrastructure, (d) the Ministry of Defence and (e) NatureScot, Natural Resources Wales and Northern Ireland Environment Agency. We requested their expenditure from the three previous financial years (i.e. 2017–2018, 2018–2019 and 2019–2020) and from this calculated an average expenditure per annum.

To compare the nectar sugar productivity (kg/ha cover/year) among study plant species, we used data from supplementary tables S11 and S13 in Baude et al. (2016).

2.4 | Statistical analysis

Statistical analyses were conducted in 'R' software (version 4.0.3; R-Project, 2021). The package VEGAN (Oksanen et al., 2007) was used to generate Shannon–Wiener diversity indices ($H' = -\sum_{i=1}^{R} p_i \ln p_i$).

For the analysis of the field study data (Figure 1), we used Generalized Linear Models (GLM) with a quasi-Poisson error structure, as the data were overdispersed. Flower-visitor abundance and Shannon–Wiener diversity indices (i.e. response variables) were analysed separately. Both models included six explanatory variables:

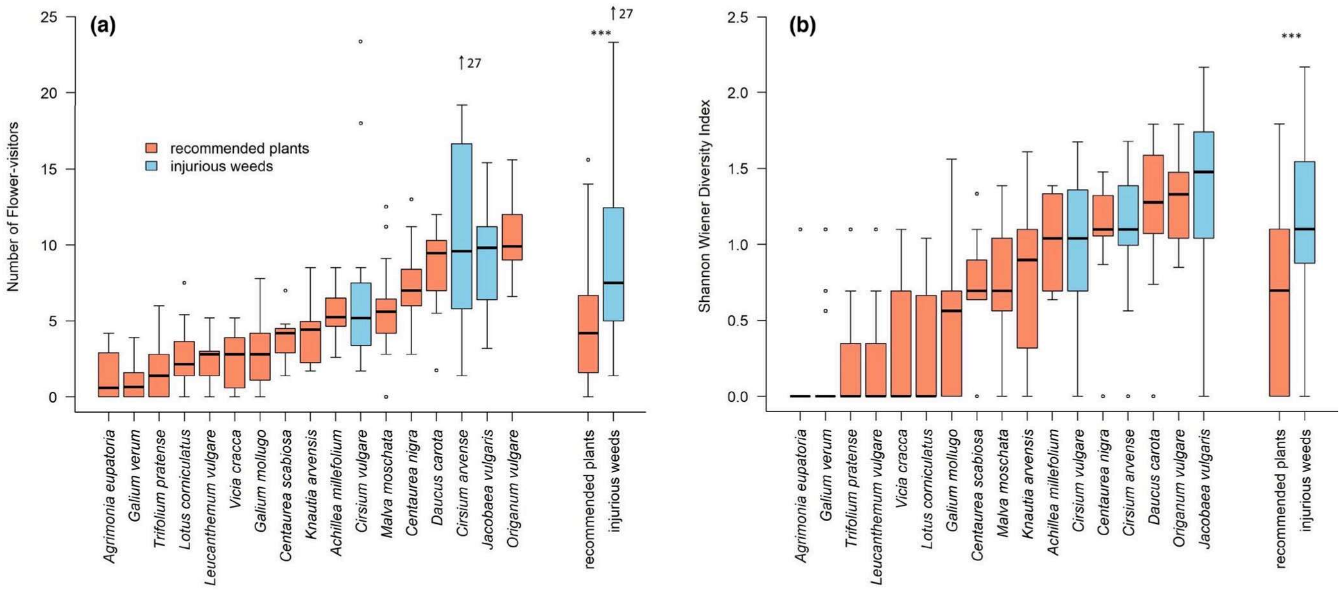

FIGURE 1 Field study results: (a) number of flower-visitors, and (b) Shannon–Wiener diversity index, recorded per 1 m^2 of our study plants during 5-min censuses. Box plots show median (horizontal line), Q1 and Q3 (boxes) and 1.5 × IQR (error bars) for 14 of the flower species recommended for pollinator-targeted agri-environmental options (orange) and the three insect-pollinated injurious weeds (blue). ***Significance level $p < 0.001$

(a) plant class (i.e. injurious weed or recommended plant species), (b) study site, (c) area (m^2) of the study plant within a 3 m radius of each patch, (d) time of day, (e) proportion cover of the study plant within each patch and (f) distance of study patch to the study site boundary. Proportion data (i.e. proportion of study plant within each patch) were logit transformed prior to analysis. Models were simplified using backward elimination of non-significant variables and model comparison using ANOVA. The data presented in Figure 1 have been adjusted to account for the proportion cover of the study plant recorded at each study patch.

For the analysis of the two databases (Figure 2), we used Linear Models (LM) when the data met the assumptions of a normal distribution (i.e. z-scores for skew and kurtosis were between −1.96 and 1.96) or GLM with a quasi-Poisson error structure. For the DoPI data, four response variables were analysed separately. These were, for each study plant species: (a) the observed number of pollinator species recorded visiting, (b) the estimated number of pollinator species recorded visiting (Chao Estimate), (c) Shannon–Wiener diversity indices and (d) the number of conservation-listed pollinator species recorded visiting. For the DBIF data, two response variables were analysed separately. These were, for each plant species: (a) the observed number of herbivorous species associated and (b) the observed number of conservation-listed herbivorous species associated. All DoPI and DBIF analysis included two explanatory variables: (a) plant class (i.e. injurious weed or recommended plant species) and (b) the geographical distribution of each plant species (number of unique UK hectads occupied). Given the likely influence of the geographical distribution of the study plant species on the associations recorded in both databases, this variable was retained in the analysis of the DoPI and DBIF data, regardless of the *p*-values returned by GLM or LM.

3 | RESULTS

3.1 | Field study: flower-visitor abundance and diversity

The three insect-pollinated injurious weeds and 14 of the recommended plant species were sufficiently abundant at more than one study site to enable 1 m^2 patches to be observed on 12–15 sampling occasions. On average, we collected data from 14.0 plant species per site (Appendix S2). In total, 226 surveys were completed. 767 insects were recorded, half of which were Hymenoptera (49%), followed by Diptera (35%), Coleoptera (11%) and Lepidoptera (5%; Appendix S3).

Overall, we recorded twice as many individual insects per 1 m^2 plant on the three insect-pollinated injurious weeds than the 14 recommended plant species (6.38 vs. 3.26; GLM; $F = 43.4$, $p < 0.001$; Figure 1). In our model of flower-visitor abundance, percentage cover of the study plant species within the study patches was significant and retained (47.56% vs. 57.35%; GLM; $F = 17.2$, $p < 0.001$). Study site, the area of the plant species within 3 m of the observation patch (2.89 m^2 vs. 2.91 m^2), time of day (13.34 vs. 13.33) and distance of the study patches to the edge of the study site (91.18 m vs. 81.62 m) were all non-significant and removed (Appendix S4).

A similar pattern was seen in our analysis of Shannon–Wiener diversity indices. Percentage cover of the study plant species within the study patch (GLM; $F = 6.0$, $p = 0.015$) and study site (GLM; $F = 2.8$, $p = 0.017$) were significant and retained. The area of the study plant within 3 m of our observation patches, time of day and distance of the study patches to the edge of the study site were

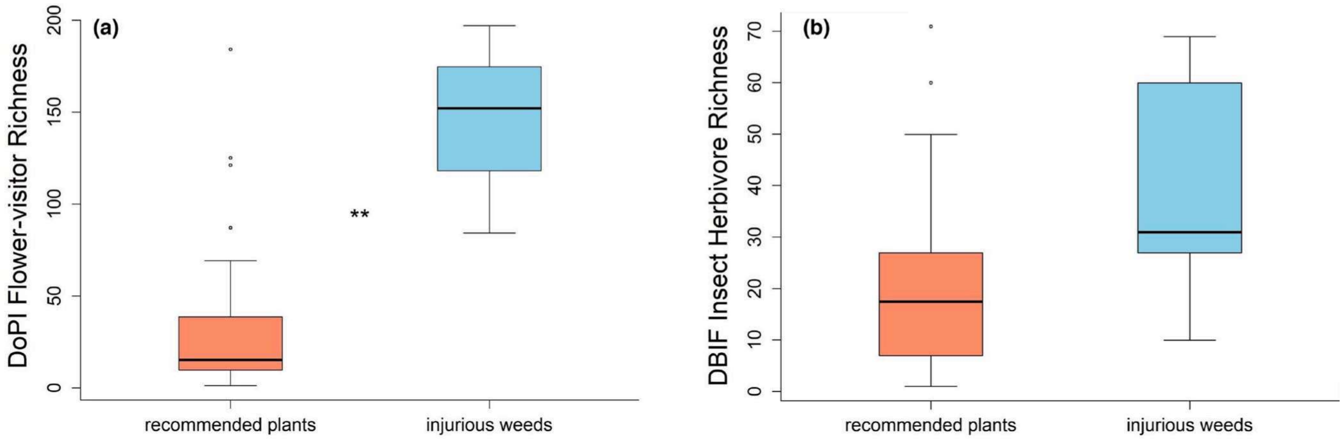

FIGURE 2 Flower-visitor species richness in the (a) Database of Pollinator Interactions (DoPI) and (b) herbivorous insect species richness in the Database of Insects and their Food Plants (DBIF) for the injurious weeds (blue) and the DEFRA recommended plants (orange). Displayed is the median (horizontal line), Q1 and Q3 (box) and 1.5 × IQR (error bars). **Significance level $p < 0.01$

all non-significant and removed (Appendix S4). The mean Shannon–Wiener diversity of the three insect-pollinated injurious weeds was greater than that of the 14 recommended plant species (1.14 vs. 0.65; GLM; $F = 25.0$, $p < 0.001$; Figure 2).

3.2 | Database of pollinator interactions (DoPI) analysis

A similar trend was apparent in the meta-analysis of the DoPI data. Four times as many insect species have been recorded visiting the three injurious weeds than the recommended plant species (144.3 vs. 33.9; LM, $F = 11.8$, $p = 0.002$; Appendix S5). This was echoed in the Chao1 estimates (338.2 vs. 76.9; LM, $F = 23.0$, $p < 0.001$), the number of conservation status insects (10.7 vs. 2.4; LM, $F = 10.1$, $p = 0.003$) and the Shannon–Wiener diversity index (3.7 vs. 2.2; LM, $F = 6.0$, $p = 0.020$). The geographical distribution (i.e. number of hectads) per plant species was significant in the analysis of the Shannon–Wiener diversity index (LM, $F = 4.7$, $p = 0.038$). However, it was found to be non-significant in the analysis of the number of observed insect species (LM, $F = 3.6$, $p = 0.067$), Choa1 estimates (LM, $F = 1.0$, $p = 0.332$) and the number of conservation-listed insect species (LM, $F = 1.5$, $p = 0.224$) associated with the study plants. Overall, for the number of observed flower-visitor species, the three insect-pollinated injurious weeds were ranked 4 (*C. arvense*), 6 (*J. vulgaris*) and 13 (*C. vulgare*) out of the 387 plant species analysed (mean rank = 7.7). Eight of the top 10 species are considered agricultural or garden weeds (Long, 1910). Across the four major UK pollinator groups (Coleoptera, Diptera, Hymenoptera and Lepidoptera), the weeds were ranked respectively: *C. arvense* (5, 3, 4 and 1), *J. vulgaris* (15, 6, 7 and 3.5) and *C. vulgare* (20, 21, 8.5 and 3.5) out of 387 plant species. For the number of conservation-listed flower-visitor species (Appendix S6), they were ranked 2 (*J. vulgaris*), 11.5 (*C. arvense*) and 81 (*C. vulgare*) out of 916 plant species (mean rank = 31.5).

3.3 | Database of insects and their food plants (DBIF) analysis

Almost twice the number of herbivorous insect species, on average, are associated with the five injurious weeds than the 33 recommended plant species (39.4 vs. 20.5). Although this difference was non-significant (LM, $F = 1.1$, $p = 0.300$), it was significant if geographical distribution was excluded from analysis (ANOVA, $F_{1,39} = 4.76$, $p = 0.035$). This suggests that more widely distributed plant species have greater herbivorous insect associations. The number of conservation status insect species associated with the injurious weeds was 40% greater than for the recommended plants species (4.2 vs. 2.9, mean per plant species), but this difference was also non-significant (GLM, $F = 0.018$, $F = 0.02$, $p = 0.895$). Geographical distribution was found to be significantly positively related to both the number of herbivorous insect species (LM, $F = 10.8$, $p = 0.002$) and the number of conservation-listed species (GLM, $F = 5.86$, $p = 0.020$) associated with the study plant species. Overall, the five injurious weed species were ranked 23 (*R. obtusifolius*), 29.5 (*J. vulgaris*), 74 (*C. arvense*), 91 (*C. vulgare*) and 239 (*R. crispus*) of the 1,033 species listed, with regard to herbivorous insect associations (mean rank = 90.6). For the number of conservation-listed insect species (Appendix S6), ranks were 27 (*J. vulgaris*), 34 (*R. obtusifolius*), 116 (*C. vulgare*), 173 (*C. arvense*) and 338.5 (*R. crispus*) of the 1,033 plant species listed (mean rank = 128.9).

3.4 | Auxiliary information: financial costs of controlling injurious weeds and nectar production

Under one third (18 of 55; 32%) of councils provided an estimate of the annual costs of controlling the five injurious weeds. Across these 18 councils, over half (10/18) reported that they specifically target ragwort, with five councils using volunteers to remove this species. Eight reported that they do not actively control any of these species. The average spent across all reporting councils was £6,406 per annum. Therefore, we estimate that the 398 UK councils spend in the region of £2.5 million per year controlling the injurious weeds. Highways England provided a figure of £754,000/year and the Northern Ireland Department of Infrastructure £6,000/year to control injurious weeds. A recent report indicates that controlling weeds on the UK rail network costs ~£7 million per year (Collings, 2017). Natural England estimates that they spend £100,000/year controlling ragwort (FOI, 2016). Natural Resources Wales and Nature Scot report that they both spend £4,000/year controlling ragwort.

The data from Baude et al. (2016) indicate that the three injurious insect-pollinated weeds produce, on average, four times the quantity of nectar sugar (1214.8 vs. 308.1 kg/ha cover/year) than do the recommended plant species. This difference was weakly significant (GLM, $F = 4.4$, $p = 0.044$). In this dataset, the three insect-pollinated injurious weed species were ranked 5 (*C. vulgare*), 12 (*J. vulgaris*) and 21 (*C. arvense*) out of the 293 plant species analysed (mean rank = 12.6). The data indicate that these three injurious weeds contributed 7.6% of the potential national nectar supply in 2007.

4 | DISCUSSION

The results of our field study indicate that the abundance and diversity of flower-visiting insects associated with the three insect-pollinated injurious weeds were twice that of recommended plants at the same localities. This finding is consistent with a meta-analysis of the data from the wider scientific literature available in the DoPI database, which indicates that fourfold more flower-visiting insect species and fourfold more conservation-listed species are associated with the injurious weeds.

Numerous other studies have identified the importance of weeds to flower-visiting insects (e.g. Feber et al., 1996; Steffan-Dewenter and Tscharntke, 2001; Kremen et al., 2002; Marshall et al., 2003; Pywell et al., 2005; Carvell, et al., 2006; Carvalheiro et al., 2011; Pocock et al., 2012; Rollin et al. 2013; Balfour et al., 2015; Requier, et al., 2015; Redhead et al., 2018; Timberlake et al., 2018). However, to our knowledge, this is the first study to directly compare the relative attractiveness of weeds versus native plants that are recommended, promoted and even incentivized for their benefits to bees, flower-visiting insects and biodiversity.

Several factors are likely responsible for this pattern. First, these three insect-pollinated injurious weeds all have generalist flowers with accessible floral rewards, meaning that they cater to a wide variety of potential flower-visiting species. Second, these plant species produce relatively high quantities of nectar sugar. Data from Baude et al. (2016) indicate that the injurious weeds produce four times more nectar than the recommended plant species per unit area of bloom. Third, the three insect-pollinated injurious weed species are geographically well-distributed (Table 1), with all five occurring in >97% of the 2,805 UK hectads. Geographical distribution is known to be related to local abundance (e.g. Brown, 1984). Furthermore, there is generally a positive relationship between the distribution of a plant species and the number of pollinators associated with it (Balfour et al., in prep).

The diversity of herbivorous insects showed a similar pattern. The five species of injurious weeds had almost twice the number of insect species associations than the recommended plants. However, this difference was not statistically significant. This was due to the positive covariation of host plant geographical distribution (Lewinsohn et al., 2005). However, there is a substantial body of evidence which points to the value of weeds to both herbivorous and predacious insects (e.g. Norris and Kogan, 2005; Barberi et al., 2010; Smith et al., 2020).

Weeds, however, have long been considered a major constraint on agricultural yields, and the aesthetics of gardens (Long, 1910). All five of the injurious weed species exhibit prodigious seed production and vigorous growth habits and can rapidly colonize disturbed habitats and out-compete existing species (Maskell et al., 2020). Their presence in crops can lead to yield losses and can reduce productivity in pastures (Harper and Wood, 1957; Cavers and Harper, 1964; Tiley, 2010). Moreover, if unchecked their dominance can lead to the loss of smaller, less vigorous species and the creation of homogeneous communities (Smart et al., 2005). Weed management in developed countries currently relies primarily on a combination of herbicide application and tillage, both of which can have negative environmental impacts (reviewed in MacLaren et al., 2020). Indeed, the general intensification of agriculture has been strongly linked to the long-term declines of both plants and associated animal groups (Robinson and Sutherland, 2002). Non-crop plants provide food for herbivores and shelter, overwintering site and reproduction (e.g. oviposition) opportunities for many species. In turn, this vegetation can host the prey of secondary consumers (e.g. carnivores). Hence, the management of the vegetation in and around fields is a major driver of biodiversity in agricultural areas. The presence of non-crop plants, such as weeds, can also have agronomic benefits, including nutrient cycling and improvement in soil physical properties (Blaix et al., 2018). They also provide resources that attract and maintain populations of parasitoids, predators and pollinators (Altieri and Letourneau, 1982; Wyss, 1995; Kleiman et al., 2020), and can make crops less apparent or less attractive to pests, thus acting as trap crops (Andow, 1991; Capinera, 2005; Castagneyrol et al., 2013; Madden et al., 2021).

Recent objections to the 1959 Weeds Act and the Ragwort Bill 2003 have mainly come from conservation organizations, such as Buglife (2014). Despite their ecological value, many common UK wildflowers that are valuable to flower-visiting and herbivorous insects are often overlooked or even disliked, as exemplified by species such as ivy (Garbuzov and Ratnieks, 2014) and bramble (Wignall et al., 2020). However, a recent petition to debate the 1959 Weeds Act in the House of Commons points to a growing awareness among the British public of their conservation utility (UKGOV, 2019). Surprisingly, the Weeds Act 1959 was never debated in parliament. The Ragwort Bill 2003 was passed on the premise that ragwort was 'on the increase' and a 'conservative estimate' was that it was responsible for 1,000 horse deaths per year (UKGOV, 2003b).

However, Countryside Survey data indicates that there was no long-term trend in ragwort occurrence between 1978 and 2007 (Laybourn et al., 2013). Likewise, the methodology employed to calculate the number of horses dying from ragwort poisoning led to a gross overestimate (e.g. Tree, 2019) with a review by the British Equine Veterinary Association concluding: ragwort toxicity is 'rare in horses and ponies' (Durham, 2014). Indeed, cattle and equines will generally not feed on ragwort unless they are forced to, for example, by being fed it dried in with hay or by poor pasture management (Wardle, 1987; Tolhurst & Oates, 2001).

The 1959 Weeds Act is from a time when agriculture was less sophisticated. Currently, British farmers spend £912 million per year on synthetic herbicides to control agricultural weeds (DEFRA, 2020b). Our data indicate that public bodies such as local councils, Natural England and Highways England are spending *c.* £10 million per year controlling injurious weeds. It is perhaps alarming that the majority of responding councils indicated that they actively control ragwort, thus classing it in the same bracket as invasive, non-native species such as Japanese knotweed *Reynoutria japonica*. Meanwhile, a further £40 m in subsidies are spent annually on planting flower species which our data indicate support less biodiversity than the injurious weeds (Rayment, 2017).

The results of this study indicate that tolerating the injurious weed species within the agricultural environment may be of greater benefit to flower-visiting insects, than the sowing of 'wildflower mixes'. These mixes are generally short-lived (Pywell et al., 2005), of non-native seed stock (Akeroyd, 1994) and generally cater to a limited suite of pollinators (e.g. Haaland et al., 2011; Scheper et al., 2013; Wood et al., 2017; Sutter et al., 2018; Gasner et al., 2021). Several studies have attempted to design strategies that reconcile agricultural productivity and biodiversity within an individual field, i.e. land sharing (e.g. Davis et al., 2012; Mézière, et al., 2015). The contrasting approach, land sparing, is where farmland is taken out of production for conservation. Current UK agricultural policy encourages neither land sparing for nor land sharing with weeds, as evidenced by the guidance provided on controlling weeds in agri-environmental areas managed for biodiversity (DEFRA, 2020c). The adoption of wildlife-friendly management is dependent on a number of factors but is largely dictated by the policy environment in which farmers operate (Jagers et al., 2018). The Environmental Land Management Scheme, to be rolled out for English farmers by the end of 2024, will replace those currently available under the EU Common Agricultural Policy. This scheme aims to reward land managers to deliver, among other environmental benefits, 'thriving plants and wildlife' (DEFRA, 2020d). Given their value to biodiversity, we hope forthcoming changes to the policy environment will provide sufficient directives and financial incentives to persuade land managers to tolerate injurious weeds. Any changes to these policies would need to consider the balance of practicality, cost (impacts on crop yields and plants of conservation concern) and benefits (effects on biodiversity, ecosystem services and direct cost savings) of tolerating weeds. To inform evidence-led policy, further work is required in these areas.

ACKNOWLEDGEMENTS

The authors thank Eva Riley, Kyle Shackleton, Tom Breeze, Louise Hutchinson, an anonymous reviewer and our funders Rowse Honey Ltd.

CONFLICT OF INTEREST

The authors have no conflict of interest.

AUTHORS' CONTRIBUTIONS

N.J.B. and F.L.W.R. conceived the ideas and designed the methodology; N.J.B. collected and analysed the data and led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

DATA AVAILABILITY STATEMENT

Data available via Figshare: https://doi.org/10.25377/sussex.18107219 (Balfour & Reitneks, 2022).

ORCID

Nicholas J. Balfour https://orcid.org/0000-0002-2426-3898
Francis L. W. Ratnieks https://orcid.org/0000-0002-3249-6325

REFERENCES

Akeroyd, J. (1994). *Seeds of destruction? Non-native wildflower seed and British floral biodiversity*. Plantlife.

Altieri, M. A., & Letourneau, D. K. (1982). Vegetation management and biological control in agroecosystems. *Crop Protection*, *1*, 405–430.

Andow, D. A. (1991). Vegetational diversity and arthropod population response. *Annual Review of Entomology*, *36*, 561–586.

Bacon, J., Jefferson, R. & Sheppard, D. (2003). Information note Common ragwort Senecio jacobaea. English Nature. Retrieved from http://publications.naturalengland.org.uk/

Balfour N. J., Castellanos, M. C., Goulson, D., Philippides, A., & Johnson, C. (in prep). *DoPI: The database of pollinator interactions*. https://www.dopi.org.uk/

Balfour, N. J., Durrant, R., Ely, A., & Sandom, C. J. (2021). People, nature and large herbivores in a shared landscape: A mixed-method study of the ecological and social outcomes from agriculture and conservation. *People and Nature*, *3*, 418–430.

Balfour, N. J., Fensome, K. A., Samuelson, E. E., & Ratnieks, F. L. (2015). Following the dance: Ground survey of flowers and flower-visiting insects in a summer foraging hotspot identified via honey bee waggle dance decoding. *Agriculture, Ecosystems & Environment*, *213*, 265–271.

Balfour, N. J., Ollerton, J., Castellanos, M. C., & Ratnieks, F. L. (2018). British phenological records indicate high diversity and extinction rates among late-summer-flying pollinators. *Biological Conservation*, *222*, 278–283.

Balfour, N. J., & Ratneiks, F. L. W. (2022). Data from: The disproportionate value of 'weeds' to pollinators and biodiversity. *Figshare*, https://doi.org/10.25377/sussex.18107219

Ball, S. G., & Morris, R. (2014). *A review of the scarce and threatened flies of Great Britain (Part 6)*. Nature Conservancy Council for England.

Ball, S. G., & Morris, R. (2015). *Britain's hoverflies: A field guide-revised and updated* (2nd ed.). Princeton University Press.

Barberi, P., Burgio, G., Dinelli, G., Moonen, A. C., Otto, S., Vazzana, C., & Zanin, G. (2010). Functional biodiversity in the agricultural landscape: Relationships between weeds and arthropod fauna. *Weed Research*, *50*, 388–401.

Bartomeus, I., Stavert, J. R., Ward, D., & Aguado, O. (2019). Historical collections as a tool for assessing the global pollination crisis. *Philosophical Transactions of the Royal Society B*, *374*, 20170389.

Baude, M., Kunin, W. E., Boatman, N. D., Conyers, S., Davies, N., Gillespie, M. A., Morton, R. D., Smart, S. M., & Memmott, J. (2016). Historical nectar assessment reveals the fall and rise of floral resources in Britain. *Nature*, *530*, 85–88.

Blaix, C., Moonen, A. C., Dostatny, D. F., Izquierdo, J., Le Corff, J., Morrison, J., Von Redwitz, C., Schumacher, M., & Westerman, P. R. (2018). Quantification of regulating ecosystem services provided by weeds in annual cropping systems using a systematic map approach. *Weed Research*, *58*, 151–164.

Brown, J. H. (1984). On the relationship between abundance and distribution of species. *The American Naturalist*, *124*, 255–279.

Buglife. (2014. *Buglife's 7 principles and 27 actions to arrest the alarming decline in UK pollinator populations*. Retrieved from https://www.buglife.org.uk/resources/publications-hub/campaigns-and-reports/

Capinera, J. L. (2005). Relationships between insect pests and weeds: An evolutionary perspective. *Weed Science*, *53*, 892–901.

Carvalheiro, L. G., Veldtman, R., Shenkute, A. G., Tesfay, G. B., Pirk, C. W. W., Donaldson, J. S., & Nicolson, S. W. (2011). Natural and within-farmland biodiversity enhances crop productivity. *Ecology Letters*, *14*, 251–259.

Carvell, C., Roy, D. B., Smart, S. M., Pywell, R. F., Preston, C. D., & Goulson, D. (2006). Declines in forage availability for bumblebees at a national scale. *Biological Conservation*, *132*, 481–489.

Castagneyrol, B., Brice, G., Christelle, P., & Hervé, J. (2013). Plant apparency, an overlooked driver of associational resistance to insect herbivory. *Journal of Ecology*, *101*, 418–429.

Cavers, P. B., & Harper, J. L. (1964). *Rumex obtusifolius* L. and *R. crispus* L. *Journal of Ecology*, *52*, 737–766.

Collings, P. 2017. *Economic impact of a glyphosate ban*. Oxford Economics. Retrieved from https://www.oxfordeconomics.com/recent-releases/The-impact-of-a-glyphosate-ban-on-the-UK-economy

Davis, A. S., Hill, J. D., Chase, C. A., Johanns, A. M., & Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PLoS ONE*, *7*, e47149.

DEFRA. (2003). *The Weeds Act 1959, Preventing the spread of harmful weeds*. Department for Environment, Food and Rural Affairs.

DEFRA. (2020a). *Biodiversity 2020: A strategy for England's wildlife and ecosystem services*. Department for Environment, Food and Rural Affairs.

DEFRA. (2020b). *Agriculture in the United Kingdom 2019*. Department for Environment, Food and Rural Affairs.

DEFRA. (2020c). *Countryside stewardship, higher tier manual*. Natural England, Forestry Commission, Department for Environment, Department for Environment, Food and Rural Affairs.

DEFRA. (2020d). *Environmental land management policy discussion document, February 2020*. Retrieved from https://consult.defra.gov.uk/elm/elmpolicyconsultation/supporting_documents/ELM%20Policy%20Discussion%20Document%20230620.pdf

Durham, A. (2014). Ragwort toxicity in the UK. *Equine Quarterly Disease Surveillance Report*, *10*, 14–19.

Falk, S. (1991a). *A review of the scarce and threatened flies of Great Britain (Part 1)*. Nature Conservancy Council for England.

Falk, S. (1991b). *A review of the scarce and threatened bees, wasps and ants of Great Britain*. Nature Conservancy Council for England.

Falk, S. J. (2015). *Field guide to the bees of Great Britain and Ireland*. British Wildlife Publishing.

Feber, R. E., Smith, H., & Macdonald, D. W. (1996). The effects on butterfly abundance of the management of uncropped edges of arable fields. *Journal of Applied Ecology*, *33*, 1191–1205.

FOI. (2016). *DEFRA Freedom of Information Request: Correspondence with public bodies relating to ragwort evidence review conducted by FERA in 2013*. https://www.whatdotheyknow.com/request/340138/response/850521/attach/html/3/RFI%208363%20response%20letter.pdf.html

Ganser, D., Albrecht, M., & Knop, E. (2021). Wildflower strips enhance wild bee reproductive success. *Journal of Applied Ecology*, *58*, 486–495.

Garbuzov, M., & Ratnieks, F. L. (2014). Ivy: An underappreciated key resource to flower-visiting insects in autumn. *Insect Conservation and Diversity*, *7*, 91–102.

Goulson, D., Nicholls, E., Botías, C., & Rotheray, E. L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, *347*, 1255957.

Haaland, C., Naisbit, R. E., & Bersier, L. F. (2011). Sown wildflower strips for insect conservation: A review. *Insect Conservation and Diversity*, *4*, 60–80.

Hadley, M. 1984. *A national review of British macro-lepidoptera*. Unpublished, Invertebrate Site Register report. Nature Conservancy Council.

Harper, J. L., & Wood, W. A. (1957). *Senecio Jacobaea* L. *Journal of Ecology*, *45*, 617–637.

Hill, M. O., Preston, C. D., & Roy, D. B. (2004). *PLANTATT-attributes of British and Irish plants: Status, size, life history, geography and habitats*. Centre for Ecology & Hydrology. Retrieved from http://nora.nerc.ac.uk/id/eprint/9535/1/PLANTATT.pdf

Hsieh, T. C., Ma, K. H., & Chao, A. (2016). iNEXT: An R package for rarefaction and extrapolation of species diversity (Hill numbers). *Methods in Ecology and Evolution*, *7*, 1451–1456.

Hyman, P. S. (1992). *A review of the scarce and threatened Coleoptera of Great Britain: Part 1*. Joint Nature Conservation Committee.

Hyman, P. S. (1994). *A review of the scarce and threatened Coleoptera of Great Britain: Part 2*. Joint Nature Conservation Committee.

Jagers, S. C., Harring, N., & Matti, S. (2018). Environmental management from left to right–on ideology, policy-specific beliefs and pro-environmental policy support. *Journal of Environmental Planning and Management*, *61*, 86–104.

Kleiman, B., Primoli, A., Koptur, S., & Jayachandran, K. (2020). Weeds, pollinators, and parasitoids-Using weeds for insect manipulation in agriculture. *Journal of Research in Weed Science*, *3*, 382–390.

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

Kremen, C., & Merenlender, A. M. (2018). Landscapes that work for biodiversity and people. *Science*, *362*(6412), 304–313.

Kremen, C., Williams, N. M., & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences of the United States of America*, *99*, 16812–16816.

Laybourn, R., Kessell, D., Jones, N., Conyers, S., Hallam, C., & Boatman, N. (2013). *Review of evidence concerning ragwort impacts, ecology and control options*. Food and Environment Research Agency.

Lewinsohn, T. M., Novotny, V., & Basset, Y. (2005). Insects on plants: Diversity of herbivore assemblages revisited. *Annual Review of Ecology, Evolution, and Systematics*, *36*, 597–620.

Long, H. C. (1910). *Common weeds of the farm and garden*. Smith, Elder & Co.

MacLaren, C., Storkey, J., Menegat, A., Metcalfe, H., & Dehnen-Schmutz, K. (2020). An ecological future for weed science to sustain crop production and the environment. *A review. Agronomy for Sustainable Development*, *40*, 1–29.

Madden, M. K., Widick, I. V., & Blubaugh, C. K. (2021). Weeds impose unique outcomes for pests, natural enemies, and yield in two vegetable crops. *Environmental Entomology*, *50*, 330–336.

Marshall, E. J. P., Brown, V. K., Boatman, N. D., Lutman, P. J. W., Squire, G. R., & Ward, L. K. (2003). The role of weeds in supporting biological diversity within crop fields. *Weed Research*, *43*, 77–89.

Maskell, L. C., Henrys, P., Pescott, O. L., & Smart, S. M. (2020). Long-term trends in the distribution, abundance and impact of native 'injurious' weeds. *Applied Vegetation Science*, *23*, 635–647.

Mézière, D., Colbach, N., Dessaint, F., & Granger, S. (2015). Which cropping systems to reconcile weed-related biodiversity and crop production in arable crops? An approach with simulation-based indicators. *European Journal of Agronomy*, *68*, 22–37.
Norris, R. F., & Kogan, M. (2005). Ecology of interactions between weeds and arthropods. *Annual Review of Entomology*, *50*, 479–503.
Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M. H. H., Solymos, P., & Wagner, H. (2007). The vegan package. *Community Ecology Package*, *10*, 631–637.
Ollerton, J., Erenler, H., Edwards, M., & Crockett, R. (2014). Extinctions of aculeate pollinators in Britain and the role of large-scale agricultural changes. *Science*, *346*, 1360–1362.
Ollerton, J., Winfree, R., & Tarrant, S. (2011). How many flowering plants are pollinated by animals? *Oikos*, *120*, 321–326.
Padovani, R., Ward, L., Smith, R. M., Pocock, M. J. O., & Roy, D. B. (2020). *Insect species richness for each plant species and insect-plant interactions from the Database of Insects and their Food Plants [DBIF] version 2*. NERC Environmental Information Data Centre. https://doi.org/10.5285/33a825f3-27cb-4b39-b59c-0f8182e8e2e4
Parsons, M. S. (1984). *A provisional national review of the status of British microlepidoptera*. Nature Conservancy Council.
Pocock, M. J., Evans, D. M., & Memmott, J. (2012). The robustness and restoration of a network of ecological networks. *Science*, *335*, 973–977.
Pywell, R. F., Warman, E. A., Carvell, C., Sparks, T. H., Dicks, L. V., Bennett, D., Wright, A., Critchley, C. N. R., & Sherwood, A. (2005). Providing foraging resources for bumblebees in intensively farmed landscapes. *Biological Conservation*, *121*, 479–494.
QGIS Development Team. (2021). QGIS Geographic Information System; Open Source Geospatial Foundation.
Rayment, M. (2017). Assessing the costs of environmental land management in the UK (A Report for the RSPB, the National Trust and The Wildlife Trusts).
Redhead, J. W., Woodcock, B. A., Pocock, M. J., Pywell, R. F., Vanbergen, A. J., & Oliver, T. H. (2018). Potential landscape-scale pollinator networks across Great Britain: Structure, stability and influence of agricultural land cover. *Ecology Letters*, *21*, 1821–1832.
Requier, F., Odoux, J. F., Tamic, T., Moreau, N., Henry, M., Decourtye, A., & Bretagnolle, V. (2015). Honey bee diet in intensive farmland habitats reveals an unexpectedly high flower richness and a major role of weeds. *Ecological Applications*, *25*, 881–890.
Robinson, R. A., & Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, *39*, 157–176.
Rollin, O., Bretagnolle, V., Decourtye, A., Aptel, J., Michel, N., Vaissière, B. E., & Henry, M. (2013). Differences of floral resource use between honey bees and wild bees in an intensive farming system. *Agriculture, Ecosystems & Environment*, *179*, 78–86.
Rowland, C., Morton, D., Carrasco Tornero, L., McShane, G., O'Neil, A., & Wood, C. (2017). *Land Cover Map 2015 (1 km percentage aggregate class, GB)*. NERC Environmental Information Data Centre.
R-Project. (2021). *R: A language and environment for statistical computing*. R Foundation. Retrieved from http://www.r-project.org/
Scheper, J., Holzschuh, A., Kuussaari, M., Potts, S. G., Rundlöf, M., Smith, H. G., & Kleijn, D. (2013). Environmental factors driving the effectiveness of European agri-environmental measures in mitigating pollinator loss – A meta-analysis. *Ecology Letters*, *16*, 912–920.
Smart, S. M., Bunce, R. G. H., Marrs, R., LeDuc, M., Firbank, L. G., Maskell, L. C., Scott, W. A., Thompson, K., & Walker, K. J. (2005). Large-scale changes in the abundance of common higher plant species across Britain between 1978, 1990 and 1998 as a consequence of human activity: Tests of hypothesised changes in trait representation. *Biological Conservation*, *124*, 355–371.
Smith, B. M., Aebischer, N. J., Ewald, J., Moreby, S., Potter, C., & Holland, J. M. (2020). The potential of arable weeds to reverse invertebrate declines and associated ecosystem services in cereal crops. *Frontiers in sustainable food systems*, *3*, 118.
Steffan-Dewenter, I., & Tscharntke, T. (2001). Succession of bee communities on fallows. *Ecography*, *24*, 83–93.
Sutter, L., Albrecht, M., & Jeanneret, P. (2018). Landscape greening and local creation of wildflower strips and hedgerows promote multiple ecosystem services. *Journal of Applied Ecology*, *55*, 612–620.
Tiley, G. E. (2010). Biological flora of the British Isles: *Cirsium arvense* (L.) scop. *Journal of Ecology*, *98*, 938–983.
Timberlake, T. P., Vaughan, I. P., & Memmott, J. (2018). Phenology of farmland floral resources reveals seasonal gaps in nectar availability for bumblebees. *Journal of Applied Ecology*, *56*, 1585–1596.
Tolhurst, S., & Oates, M. (2001). *A guide to animal welfare in nature conservation grazing. The Grazing Animals Project*. English Nature.
Tree, I. (2019). *Wilding: The return of nature to a British farm*. Pan Macmillan.
UKGOV. (1959). *Weeds Act 1959*. Retrieved from https://www.legislation.gov.uk/ukpga/Eliz2/7-8/54
UKGOV. (2003a). *Ragwort Control Act 2003*. Retrieved from https://www.legislation.gov.uk/ukpga/2003a/40/contents
UKGOV. (2003b). *Ragwort Control Bill second reading in the House of Commons*. Retrieved from https://hansard.parliament.uk/Commons/2003b-03-21/debates/1acd05c5-5e0c-4072-92ff-a83772bcb7c1/EquineWelfare(RagwortControl)Bill
UKGOV. (2019). *Petition Repeal the archaic Weeds Act 1959 to benefit pollinators and wider biodiversity*. Retrieved from https://petition.parliament.uk/archived/petitions/266743
Unwin, D. M. (1984). *A key to the families of British Diptera*. Field Studies Council.
Wardle, D. A. (1987). The ecology of ragwort (*Senecio jacobaea* L.) – A review. *New Zealand Journal of Ecology*, *10*, 67–76.
Wignall, V. R., Arscott, N. A., Nudds, H. E., Squire, A., Green, T. O., & Ratnieks, F. L. (2020). Thug life: Bramble (*Rubus fruticosus* L. agg.) is a valuable foraging resource for honeybees and diverse flower-visiting insects. *Insect Conservation and Diversity*, *13*, 543–557.
Wood, T. J., Holland, J. M., & Goulson, D. (2017). Providing foraging resources for solitary bees on farmland: Current schemes for pollinators benefit a limited suite of species. *Journal of Applied Ecology*, *54*, 323–333.
Wyss, E. (1995). The effects of weed strips on aphids and aphidophagous predators in an apple orchard. *Entomologia Experimentalis et Applicata*, *75*, 43–49.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

How to cite this article: Balfour, N. J., & Ratnieks, F. L. (2022). The disproportionate value of 'weeds' to pollinators and biodiversity. *Journal of Applied Ecology*, 59, 1209–1218. https://doi.org/10.1111/1365-2664.14132