UK Biodiversity Indicators 2021

This document supports
D1c. Status of pollinating Insects

Technical background document:

Gary D. Powney, Colin A. Harrower, Louise Barwell, Charlotte Outhwaite, Nick J.B. Isaac

For further information on D1c. Status of pollinating insects visit http://www.jncc.gov.uk/ukbi-D1c

For further information on the UK Biodiversity Indicators visit http://www.jncc.gov.uk/ukbi

D1c - Biodiversity and Ecosystem Services – status of pollinating insects – technical background document

Gary D. Powney, Colin A. Harrower, Louise Barwell, Charlotte Outhwaite, Nick J. B. Isaac

Introduction

Pollination is a vital ecosystem service that benefits agricultural and horticultural production, and is essential for maintaining wild flower biodiversity. By improving the yield, quality and resilience of crops, insect pollination has been valued at £400 million per year to the UK economy (POST, 2010). Thirty five percent of the world's agricultural output, by volume, consists of 87 crop types that benefit from pollination by animals (insects, birds and mammals), but because most of these crops are not entirely dependent on animal pollination, the amount of production directly attributable to animals is lower than this value (Klein *et al.*, 2007). There is growing concern regarding the population status of insect pollinators, and in turn the pollination service they provide (Potts *et al.*, 2010; Garratt *et al.*, 2014). As with most other areas of biodiversity, the main threats to pollinators include habitat loss, environmental pollution, climate change and the spread of alien species (Klein *et al.*, 2007; Potts *et al.*, 2010; Vanbergen & The Insect Pollinators Initiative 2013). The widespread application of pesticides is also perceived as a major threat to pollinator diversity (Brittain *et al.*, 2010). In order for governments to act upon these threats they need robust metrics on the national-scale status of pollinators and pollination, though deriving such a metric has previously been limited by the availability of suitable data and analytical techniques, and the species considered to be wild pollinators are subject to debate (Hutchinson *et al.* 2021). With the increase in citizen science, the availability of large-scale biological record data has increased (Silvertown, 2009). Such data are collected without a standardized survey protocol and therefore extracting reliable trends from them can be difficult. However, with recent analytical advances it is now possible to estimate reliable trends from such data (van Strien *et al.*, 2013; Isaac *et al.*, 2014).

Methods

Data sources

Occurrence records of bee and hoverfly species within 1km grid cells in the UK originate from the Bees, Wasps and Ants Recording Society (BWARS) and the Hoverfly Recording Scheme biological records databases. The time-period used for the indicator was 1980 to 2017, as this represents a core period of recording for these taxa in the UK. Bee species were filtered (following expert guidance from BWARS) so that only species considered to be wild pollinators were included. Species that had undergone taxonomic changes or had taxonomic issues during the time frame of the indicator were excluded from the analysis. Furthermore, models based on species with less than 50 records tend to be unreliable (Powney *et al.*, 2019), and were therefore excluded from the analysis. The final composite indicator was based on 377 species of wild pollinator, see Appendix 1 for a list of species covered. Note that the species considered to be wild pollinators are subject to review, following feedback from the scientific community and the publication of a literature review of field survey data recording wild bee visits to crops in Great Britain and Europe (Hutchinson et al. 2021).

Generating species' occupancy models

The data used to produce the indicator were not collected using a standardised protocol, but instead are a collation of unstructured biological observations collected by a large network of volunteer recorders. Such data tend to contain many forms of sampling bias and noise, making it hard to detect genuine signals of change (Tingley & Beissinger, 2009; Hassall & Thompson, 2010; Isaac et al., 2014). Recent studies have highlighted the value of Bayesian occupancy models for estimating species occurrence in the presence of imperfect detection (van Strien et al., 2013; Isaac et al., 2014). This approach uses two hierarchically coupled sub-models: an occupancy sub-model (i.e. presence verses absence), and a detection sub-model (i.e. detection verses non-detection).

Together these sub-models estimate the conditional probability that a species is detected when present. Species-specific time series estimates are derived from a Bayesian occupancy model, described in Outhwaite et al. (2019) and following van Strien et al. (2013) and Isaac et al. (2014),

2

with improvements based on Outhwaite et al. (2018). Annual estimates of occupancy, with estimates of uncertainty, are available for 5,293 UK invertebrate, bryophyte and lichen species for the period 1970 to 2015 (Outhwaite et al. 2019). These models are updated as and when new data become available from recording schemes. For each site-year combination the model estimates presence or absence for the species in question given variation in detection probability: from this the proportion of occupied sites ('occupancy') was estimated for each year.

Species-specific trends

For each species, the long- and short-term trend in occupancy was estimated as the mean annual percent change (over the time-period in question) across 1,000 estimates from the posterior distribution. Species were grouped into 1 of 5 categories based on both their short-term and long-term occupancy trend (Table 1). The threshold values for each category were based on those of the wild bird indicator; whether an individual species is increasing or decreasing has been decided by its rate of annual change over the time period (long or short) of interest. If the rate of annual change would lead to an occupancy increase or decrease of between 25% and 49% over 25 years, the species is said to have shown a 'weak increase' or a 'weak decline' respectively. If the rate of annual change would lead to a population increase or decrease of 50% or more over 25 years, the species is said to have shown a 'strong increase' or a 'strong decline', respectively. These thresholds are used in the Birds of Conservation Concern status assessment for birds in the UK.

Table 1: Thresholds used to define individual species trends

Category	Thresholds	Threshold – equivalent
Strong increase	Above +2.81% per annum	+100% over 25 years
Weak increase	Between +1.16% and +2.81% p.a.	+33% to +100% over 25 years
Stable	Between -1.14 % and +1.16% p.a.	-25% to +33% over 25 years
Weak decrease	Between -2.73% and -1.14% p.a.	-50% to -25% over 25 years
Strong decrease	Below -2.73% p.a.	-50% over 25 years

Asymmetric percentage change thresholds are used to define these classes as they refer to proportional change, where a doubling of a species index (an increase of 100%) is counterbalanced by a halving (a decrease of 50%).

Filtering out reliable occupancy estimates

Not all model outputs can be considered reliable. A set of rules based on method exploration and testing were therefore employed to determine which index values were suitable for inclusion in a composite indicator. Species with less than 50 total records were excluded from the composite indicator, removing the rare species for which trends in distribution could not be reliability estimated. To further improve reliability, species with a gap in records greater than 10 consecutive years were excluded. In addition, to ensure a reasonable time series was available, species required at least 10 years of occupancy estimates for inclusion. Each individual species' occupancy estimates were clipped to after the first detection for the species in question.

Creating a composite indicator

Most species-based biodiversity indicators calculate the composite index as the geometric mean of indices for those species that contribute data in that year, relative to a value of 100 in the starting year.

$$I_t = 100.(\Pi\{x_{1,t}, x_{2,t} .. x_{n,t}\})^{1/n} / I_1 \qquad \text{Eq 1}$$

Where I_t is the value of the indicator in year t and $x_{1,t}$ is the index value for species 1 in year t. Under this approach, the proportional change in the indicator from one year to the next, Δ_t, is mathematically equivalent to the geometric mean growth rate from years t-1 to t.

$$\Delta_t = I_t/I_{(t-1)} = (\Pi\{\lambda_{1,t}, \lambda_{2,t} .. \lambda_{n,t}\})^{1/n} \qquad \text{Eq 2}$$

$$\lambda_{i,t} = x_{i,t} / x_{i,(t-1)} \qquad \text{Eq 3}$$

Where $\lambda_{i,t}$ is the growth rate for species i from year t-1 to t, and n is the number of species with index values in both year t-1 and t. The geometric mean is appropriate for indices based on abundance data, which is bounded at zero but unbounded above. However, occupancy estimates are bounded at both zero and one (a species cannot occupy more than 100% of available sites). To retain this property in the indicator, the arithmetic mean of the change in log odds was selected as an appropriate statistic, thus

$$\Delta_t = I_t/I_{(t-1)} = \Sigma\{\gamma_{1,t}, \gamma_{2,t} .. \gamma_{n,t}\}/n \qquad \text{Eq 4}$$

$$\gamma_{i,t} = \log(p_{i,t}/(1-p_{i,t})) - \log(p_{i,(t-1)}/(1-p_{i,(t-1)})) \qquad \text{Eq 5}$$

Where $p_{i,t}$ is the proportion of occupied sites (i.e. "occupancy") for species i in year t, and $\gamma_{i,t}$ is the log of the growth rate in the odds of the average site being occupied by species i between years t-1 and t. Following convention, the headline indicator is set to start at 100 with a lower bound of zero:

$$I_t = 100 * \exp(\Sigma\{\Delta_1, \Delta_2 .. \Delta_t\}) \qquad \text{Eq 6}$$

Reformulating the composite indicator in terms of growth rates has 2 distinct advantages over the conventional approach to constructing indicators. First, it means that the categorisation of species as 'increasing' or 'decreasing' can be made from the same set of data (the growth rates) as the construction of the headline indicator. Second, it provides an elegant solution to the problem of species that join the indicator after the first year (i.e. where the first year is unreliable): other indicators typically adopt a complicated rescaling approach to ensure that species entering the indicator after the first year do not bias the overall assessment. It also makes a simple and robust, though untestable, assumption about species that drop out of the indicator prior to the final year: specifically, it assumes that their fluctuations are the same, in aggregate, as those of the species that remain in the indicator.

Assessing the indicator line

As noted above, the index values are not classical point estimates, but rather derived from a posterior distribution of values. This distribution makes it possible to incorporate uncertainty in the annual occupancy estimates formally around the indicator line. Since each γ in equation 4 has a distribution of 10,000 values, it is trivial to calculate Δt and It as a distribution, the quantiles of which measure the credible intervals of the indicator. Since this approach leads to full propagation of uncertainty from the results of each species' model, the magnitude of uncertainty around the indicator line is expected to be larger than for other indicators, where the index values are assumed to be known without error and the uncertainty is estimated via bootstrapping. In Bayesian statistics, the magnitude of uncertainty around parameter estimates is generally referred to as the 'credible intervals', as opposed to the 'confidence intervals' derived from classical frequentist statistics. For many applications, the Bayesian credible intervals and Frequentist confidence intervals are very similar. However, the interpretation is quite different, reflecting differences in the underlying philosophy of the 2 statistical paradigms. In Frequentist statistics, the 95% confidence intervals suggest that, if an experiment were repeated many times, the true value of the parameter (e.g. the index in the most recent year) would fall within the intervals 95% of the time. In other words, the uncertainty is an expression about the data collection process, and the parameter is assumed to be fixed. By contrast, the Bayesian approach treat the data (i.e. the species' data) as fixed and expresses uncertainty in terms of the parameter being estimated (whilst accounting for uncertainty in the data). The credible intervals around a Bayesian indicator reflect the probability that the indicator value lies within those intervals. The 90% credible intervals were chosen for making the short- and long-term assessment of a trend in the indicator line. Thus if the upper limit of the 90% credible interval falls below 100, this gives at least 95% probability (not 90%), based on the posterior distribution, of the index having declined. Thus, the long-term assessment is a simple test of whether the value 100 lies inside or outside the 90% credible intervals for the focal year. Similarly, the short-term assessment tests whether the median

value in year t-5 lies within the 90% credible intervals for the focal year, t.

Results

The indicator for all pollinators and for hoverflies has not been updated since the 2020 publication. The wild bee indicator has been extended by one additional year to 2019.

- The indicator (Figure 1) shows the average relative change in the area over which each of 377 species of pollinator was found, as measured by the number of 1km grid squares across the UK in which they were recorded – this is referred to as the 'occupancy index'.

- Over the long term (1980 to 2017), the pollinator indicator showed 30% decline, and was therefore assessed as declining.

- Temporal patterns of change in the pollinator indicator showed a steady decline from 1987 onwards.

- Between 2012 and 2017 the indicator decreased by approximately 2%, therefore the short-term trend was assessed as little change.

- Over the long term, 19% of pollinator species became more widespread (7% showed a strong increase), and 49% became less widespread (24% showed a strong decrease).

- By contrast, over the short term, a greater proportion of species were increasing (46%; with 34% exhibiting a strong increase) than decreasing (43%; with 36% exhibiting a strong decrease).

- As individual pollinator species become more or less widespread, the communities in any given area become more or less diverse, and this may have implications for pollination as more diverse communities are, in broad terms, more effective in pollinating a wide range of crops and wild flowers.

The indicator plot was also produced for the bee (Figure 2) and hoverfly (Figure 3) species separately. Note that available data permits calculation of the indicator up to 2019 for bees, but the headline (bees and hoverflies) and hoverfly indicators only run to 2017.

The wild bee index fluctuates around its initial value over much of the time-series. In 1983,1985, 1997, 2006 and 2015, there is some evidence of a recovery. However, the bee index in 2019 was estimated to be 9% lower than in 1980. and is assessed as decreasing (Figure 2). A larger proportion of bee species had decreased than increased over the long term (37% decreased and 24% increased). Over the short term, the number of species decreasing (40%) and increasing (42%) is similar. The recent decline in bees is striking. A run of wet summers and agricultural practices, including pesticide use, have both been implicated in the recent decline in bees (Stanley et al., 2015). However, further research is needed to better understand the relative importance of these potential drivers of change.

With regard to hoverflies, the index was at a peak in 1987 (109% compared to its 1980 value), and then (apart from some minor increases), underwent a progressive decline. Thus, the indicator is approximately 41% lower in 2017 than in 1980. Over the short term (2012 to 2017), the indicator decreases by just over 3%. A greater proportion of hoverflies have declined than increased in occupancy over both the long and short term (1980 to 2017: 55% decreased and 15% increased; 2012 to 2017: 49% decreased and 44% increased). It is not clear why hoverflies show a different trend to bees, although differences in the life cycle will mean they respond differently to weather events and habitat change.

Figure 1: Change in the distribution of wild pollinators (n = 377) in the UK between 1980 and 2017. The shaded region is the 90% credible intervals of the annual occupancy estimates and represents the uncertainty surrounding the annual estimates. The solid line illustrates the rescaled indicator value. The proportion of pollinator species in each trend category is based on the mean annual change in occupancy over both a) the long term (1980 to 2017) and b) the short term (2012 to 2017).
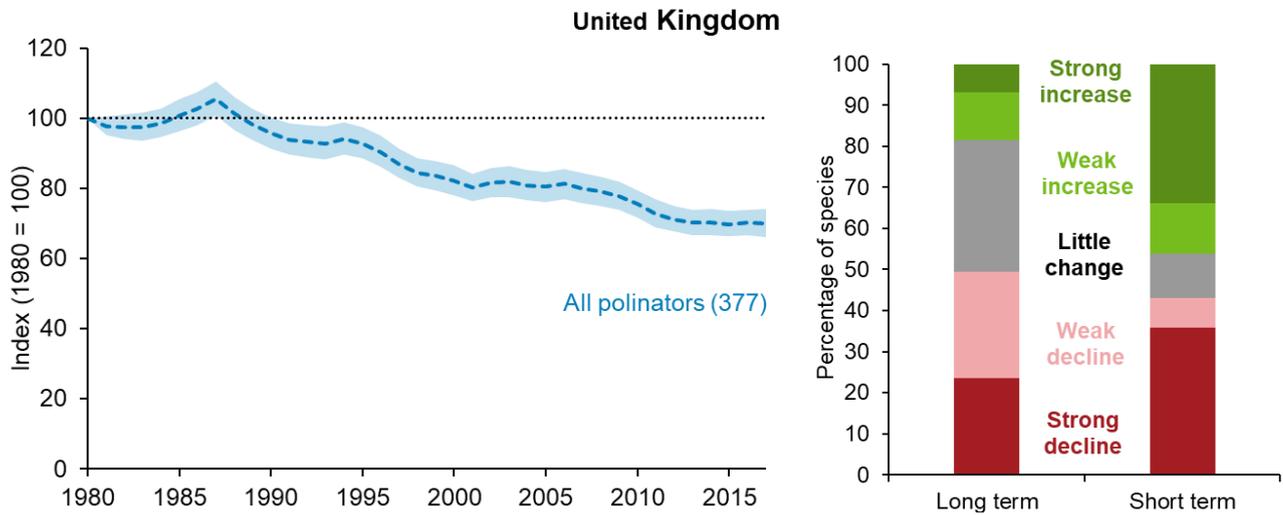

Figure 2: Change in the distribution of pollinating wild bee species (n = 148) in the UK between 1980 and 2019. The shaded region is the 90% credible intervals of the annual occupancy estimates and represents the uncertainty surrounding the annual estimates. The solid line illustrates the rescaled indicator value. The proportion of pollinating wild bee species in each trend category is based on the mean annual change in occupancy over both a) the long term (1980 to 2019) and b) the short term (2014 to 2019).
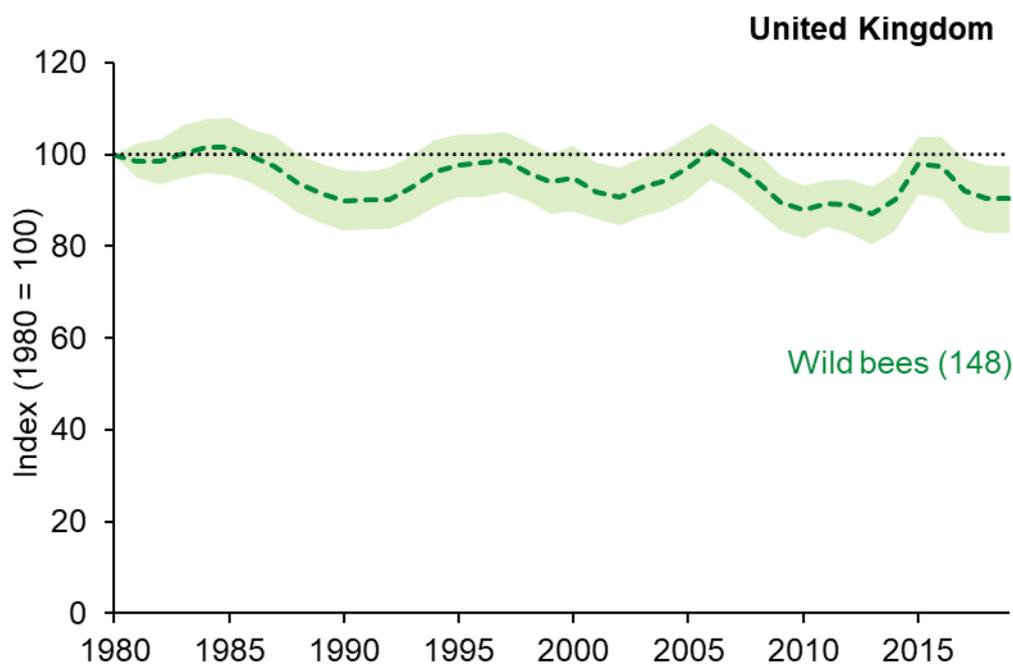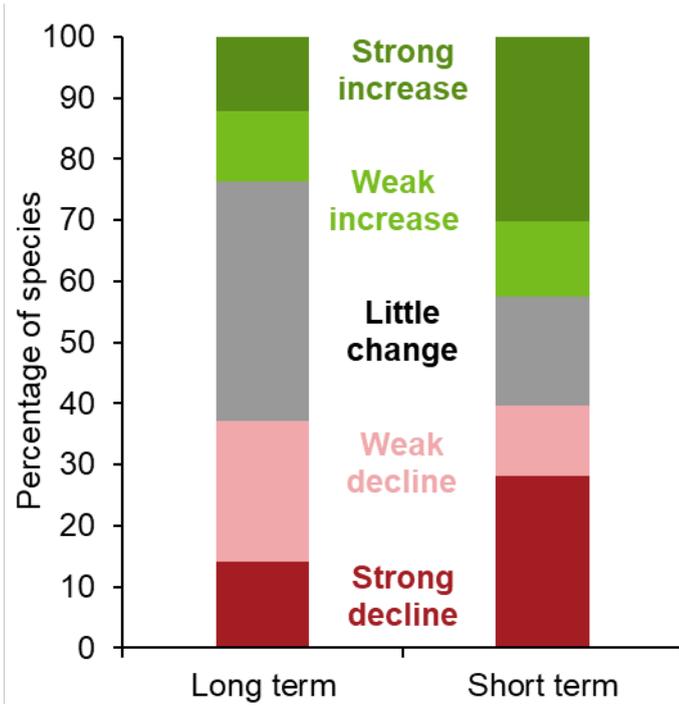
Figure 3: Change in the distribution of hoverfly species (n = 229) in the UK between 1980 and 2017. The shaded region is the 90% credible intervals of the annual occupancy estimates and represents the uncertainty surrounding the annual estimates. The solid line illustrates the rescaled indicator value. The proportion of hoverfly species in each trend category is based on the mean annual change in occupancy over both a) the long term (1980 to 2017) and b) the short term (2012 to 2017).
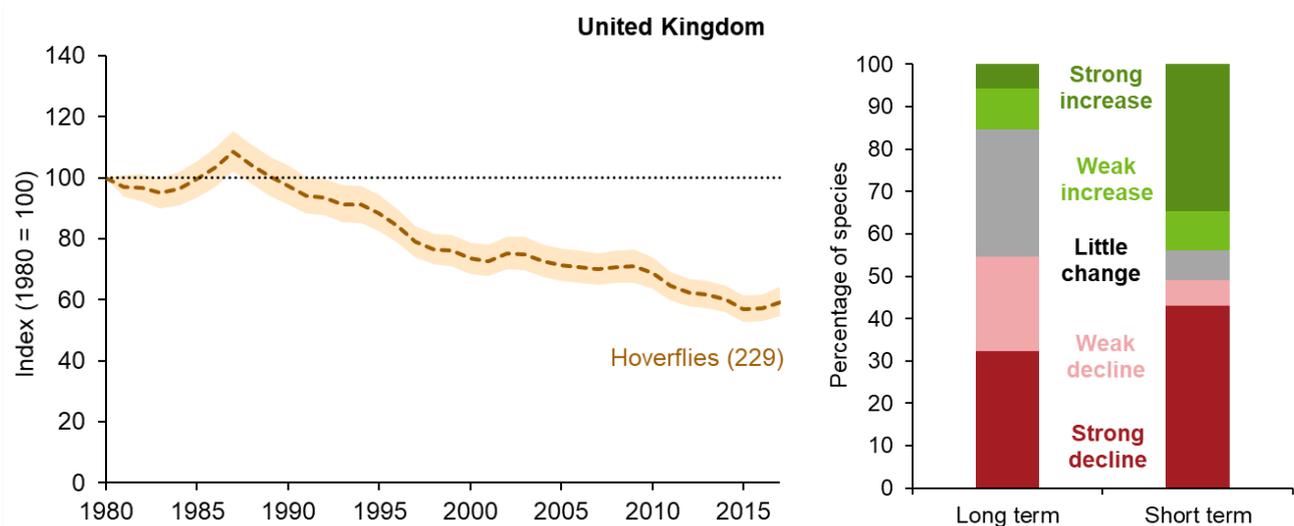

Additional Note on The Indicator.

Species only contribute to the indicator from the year in which they were first recorded, until the year in which they were last recorded. This means that many species only contribute for a subset of years, and hence the number of species contributing varies between years. Figure 4 shows the number of species contributing to the pollinator indicator per year up to 2018. Note that the statistical method used to produce the indicator must treat the number of species in the first year, 1980, as zero.

In 2019, the indicator spanned the period 1980 to 2016. As more data has become available for wild

bees we revisited this in 2020 to decide whether the indicator may be extended. The number of species contributing and the precision of the indicator in a given year are related; generally speaking, the more species, the better the precision. Therefore, as data becomes more sparse towards the end of the time-series (and hence fewer species contribute), we can use these measures to decide on an appropriate cut-off (final year) for the indicator. Based on the dramatic drop in number of species in 2018 – largely because there are no data for hoverflies - we do not include this year in the indicator. While there is a small drop in the number of species from 2016 to 2017 (Figure 4), the precision of the indicator is similar (Figure 1), so we decided that it is justifiable for 2017 to be the final year in the indicator time series.

Figure 4. Number of species contributing to the indicator in each year. Note that the statistical method used to generate the indicator requires that the start year, 1980, must be treated as having zero species.
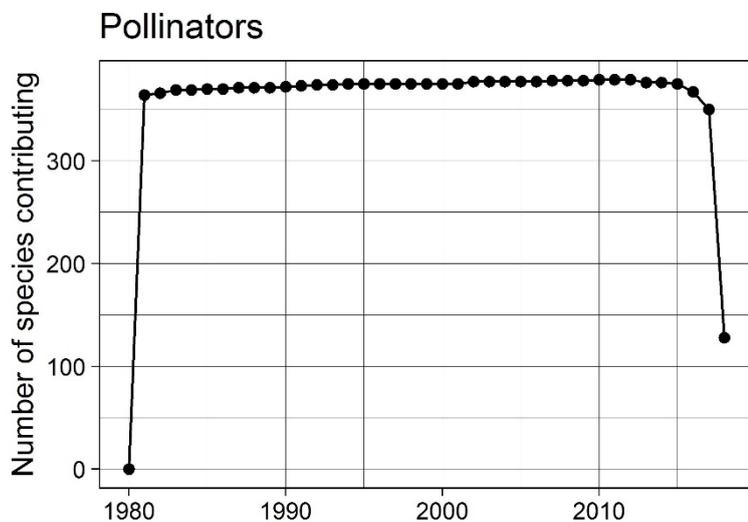
References

Breeze, T.D., Bailey, A.P., Balcombe, K.G. & Potts, S.G. (2011). Pollination services in the UK: How important are honeybees? *Agriculture, Ecosystems & Environment*, **142**, 137–143.

Brittain, C. A., Vighi, M., Bommarco, R., Settele, J. & Potts, S.G. (2010). Impacts of a pesticide on pollinator species richness at different spatial scales. *Basic and Applied Ecology*, **11**, 106– 115.

Garratt, M.P.D., Truslove, C.L., Coston, D.J., Evans, R.L., Moss, E.D., Dodson, C., Jenner, N., Biesmeijer, J.C. & Potts, S.G. (2014). Pollination deficits in UK apple orchards. *Journal of Pollination Ecology*, **12**, 9–14.

Hassall, C. & Thompson, D.J. (2010). Accounting for recorder effort in the detection of range shifts from historical data. *Methods in Ecology and Evolution*, **1**, 343–350.

Hutchinson, L.A., Oliver, T.H., Breeze, T.D., Bailes, E.J., Brünjes, L., Campbell, A.J., Garratt, M.P.D. (2021). Using ecological and field survey data to establish a national list of the wild bee pollinators of crops. Agriculture, Ecosystems & Environment, 315, 107447.

Isaac, N.J.B., van Strien, A.J., August, T.A., de Zeeuw, M.P. & Roy, D.B. (2014). Statistics for citizen science: extracting signals of change from noisy ecological data. *Methods in Ecology and Evolution*, **5**, 1052–1060.

Klein, A.-M., Vaissière, 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. Biological sciences / The Royal Society*, **274**, 303–13.

Outhwaite, C.L., Chandler, R.E., Powney, G.D., Collen, B., Gregory, R.D. & Isaac N. J. B. (2018). Prior specification in Bayesian occupancy modelling improves analysis of species occurrence data. *Ecological Indicators*, **93**, 333-343

Outhwaite, C.L., Powney, G.D., August, T.A., Chandler, R.E., Rorke, S., Pescott, O.L., … Isaac, N.J.B. (2019). Annual estimates of occupancy for bryophytes, lichens and invertebrates in the UK, 1970–2015. *Scientific Data* **6**, 259.

POST (2010) *Insect Pollination*, London.

Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O. & Kunin, W.E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in ecology & evolution*, **25**, 345–53.

Powney, G.D., Carvell, C., Edwards, M., Morris, R.K.A., Roy, H.E., Woodcock, B.A. & Isaac, N.J.B. (2019). Widespread losses of pollinating insects in Britain. *Nature Communications*, **10**, 1018.

Silvertown, J. (2009). A new dawn for citizen science. *Trends in ecology & evolution*, **24**, 467–471.

Stanley, D.A. Garratt, M.P.D., Wickens, J.B., Wickens, V.J., Potts, S.G. & Raine, N.E. (2015). Neonicotinoid pesticide exposure impairs crop pollination services provided by bumblebees. *Nature*, online early.

Tingley, M.W. & Beissinger, S.R. (2009). Detecting range shifts from historical species occurrences: new perspectives on old data. *Trends in Ecology & Evolution*, **24**, 625–633.

Van Strien, A.J., van Swaay, C. A. M. & Termaat, T. (2013). Opportunistic citizen science data of animal species produce reliable estimates of distribution trends if analysed with occupancy models. *Journal of Applied Ecology*, **50**, 1450–1458.

Vanbergen, A.J. & The Insect Pollinators Initiative. (2013). Threats to an ecosystem service: pressures on pollinators. *Frontiers in Ecology and the Environment*, **11**, 251–259.

Woodcock, B.A., Edwards, M., Redhead, J., Meek, W.R., Nuttall, P., Falk, S., Nowakowski, M. & Pywell, R.F. (2013). Crop flower visitation by honeybees, bumblebees and solitary bees: Behavioural differences and diversity responses to landscape. *Agriculture, Ecosystems & Environment*, **171**, 1–8.

Appendix : The list of the 377 species included in the pollinator indicator.

Bees	Hoverflies
Andrena alfkenella	Anasimyia contracta
Andrena angustior	Anasimyia interpuncta
ndrena apicata	Anasimyia lineata
Andrena argentata	Anasimyia lunulata
Andrena barbilabris	Anasimyia transfuga
Andrena bicolor	Arctophila superbiens
Andrena bimaculata	Brachyopa bicolor
Andrena bucephala	Brachyopa insensilis
Andrena chrysosceles	Brachyopa pilosa
Andrena cineraria	Brachyopa scutellaris
Andrena clarkella	Brachypalpoides lentus
Andrena coitana	Brachypalpus laphriformis
Andrena congruens	Caliprobola speciosa
Andrena denticulata	Callicera rufa
Andrena dorsata	Chalcosyrphus nemorum
Andrena falsifica	Chamaesyrphus scaevoides
Andrena ferox	Cheilosia albipila
Andrena flavipes	Cheilosia antiqua
Andrena florea	Cheilosia barbata
Andrena fucata	Cheilosia bergenstammi
Andrena fulva	Cheilosia carbonaria
Andrena fulvago	Cheilosia chrysocoma
Andrena fuscipes	Cheilosia cynocephala
Andrena gravida	Cheilosia fraterna
Andrena haemorrhoa	Cheilosia griseiventris
Andrena hattorfiana	Cheilosia grossa
Andrena helvola	Cheilosia illustrata
Andrena humilis	Cheilosia impressa
Andrena labialis	Cheilosia latifrons
Andrena labiate	Cheilosia longula
Andrena lapponica	Cheilosia mutabilis
Andrena marginate	Cheilosia nebulosa
Andrena minutula	Cheilosia nigripes
Andrena minutuloides	Cheilosia pagana
Andrena nigriceps	Cheilosia proxima
Andrena nigroaenea	Cheilosia pubera
Andrena nitida	Cheilosia scutellata
Andrena nitidiuscula	Cheilosia semifasciata
Andrena niveata	Cheilosia soror
Andrena ovatula	Cheilosia variabilis
Andrena praecox	Cheilosia velutina
Andrena proxima	Cheilosia vernalis
Andrena rosae	Cheilosia vicina
Andrena ruficrus	Cheilosia vulpina
Andrena scotica	Chrysogaster cemiteriorum
Andrena semilaevis	Chrysogaster solstitialis
ndrena similis	Chrysogaster virescens
Andrena simillima	Chrysotoxum arcuatum

Andrena subopaca
Andrena synadelpha
Andrena tarsata
Andrena thoracica
Andrena tibialis
Andrena trimmerana
Andrena vaga
Andrena varians
Andrena wilkella
Anthidium manicatum
Anthophora bimaculata
Anthophora furcata
Anthophora plumipes
Anthophora quadrimaculata
Anthophora retusa
Bombus distinguendus
Bombus hortorum
Bombus humilis
Bombus hypnorum
Bombus jonellus
Bombus lapidarius
Bombus muscorum
Bombus pascuorum
Bombus pratorum
Bombus ruderarius
Bombus ruderatus
Bombus soroeensis
Bombus sylvarum
Bombus terrestris
Ceratina cyanea
Chelostoma campanularum
Chelostoma florisomne
Colletes cunicularius
Colletes daviesanus
Colletes floralis
Colletes fodiens
Colletes halophilus
Colletes hederae
Colletes marginatus
Colletes similis
Colletes succinctus
Dasypoda hirtipes
Eucera longicornis
Halictus confuses
Halictus rubicundus
Halictus tumulorum
Heriades truncorum
Hoplitis claviventris
Lasioglossum albipes
Lasioglossum angusticeps
Lasioglossum brevicorne

Chrysotoxum bicinctum
Chrysotoxum cautum
Chrysotoxum elegans
Chrysotoxum festivum
Chrysotoxum verralli
Criorhina asilica
Criorhina berberina
Criorhina floccosa
Criorhina ranunculi
Dasysyrphus albostriatus
Dasysyrphus friuliensis
Dasysyrphus tricinctus
Dasysyrphus venustus
Didea fasciata
Didea intermedia
Doros profuges
Epistrophe diaphana
Epistrophe eligans
Epistrophe grossulariae
Epistrophe nitidicollis
Episyrphus balteatus
Eriozona syrphoides
Eristalinus aeneus
Eristalinus sepulchralis
Eristalis abusivus
Eristalis arbustorum
Eristalis horticola
Eristalis intricarius
Eristalis pertinax
Eristalis rupium
Eristalis tenax
Eumerus funeralis
Eumerus ornatus
Eumerus sabulonum
Eumerus strigatus
Eupeodes corollae
Eupeodes latifasciatus
Eupeodes luniger
Eupeodes nielseni
Eupeodes nitens
Ferdinandea cuprea
Ferdinandea ruficornis
Helophilus hybridus
Helophilus pendulus
Helophilus trivittatus
Heringia heringi
Lejogaster metallina
Lejogaster tarsata
Lejops vittatus
Leucozona glaucia
Leucozona laternaria

Lasioglossum calceatum
Lasioglossum cupromicans
Lasioglossum fratellum
Lasioglossum fulvicorne
Lasioglossum laevigatum
asioglossum laticeps
Lasioglossum lativentre
Lasioglossum leucopus
Lasioglossum leucozonium
Lasioglossum malachurum
Lasioglossum minutissimum
Lasioglossum morio
Lasioglossum nitidiusculum
Lasioglossum parvulum
Lasioglossum pauperatum
Lasioglossum pauxillum
Lasioglossum prasinum
Lasioglossum punctatissimum
Lasioglossum puncticolle
Lasioglossum quadrinotatum
Lasioglossum rufitarse
Lasioglossum semilucens
Lasioglossum smeathmanellum
Lasioglossum villosulum
Lasioglossum xanthopus
Lasioglossum zonulum
Macropis europaea
Megachile centuncularis
Megachile circumcincta
Megachile leachella
Megachile ligniseca
Megachile maritima
Megachile versicolor
Megachile willughbiella
Melitta haemorrhoidalis
Melitta leporine
Melitta tricincta
Osmia aurulenta
Osmia bicolor
Osmia bicornis
Osmia caerulescens
Osmia leaiana
Osmia parietina
Osmia pilicornis
Osmia spinulosa
Osmia uncinate
Osmia xanthomelana
Panurgus banksianus
Panurgus calcaratus
elangyna compositarum/labiatarum
Melanogaster aerosa

Leucozona lucorum
Mallota cimbiciformis
Melangyna arctica
Melangyna cincta
Melangyna compositarum
Melangyna labiatarum
Melangyna lasiophthalma
Melangyna quadrimaculata
Melangyna umbellatarum
Melanogaster hirtella
Melanostoma dubium
Melanostoma mellinum
Melanostoma scalare
Meliscaeva auricollis
Meliscaeva cinctella
Merodon equestris
Microdon analis
Microdon devius
Myathropa florea
Myolepta dubia
Neoascia geniculata
Neoascia interrupta
Neoascia obliqua
Neoascia podagrica
Neoascia tenur
Orthonevra brevicornis
Orthonevra geniculata
Orthonevra nobilis
Paragus haemorrhous
Parasyrphus annulatus
Parasyrphus lineola
Parasyrphus malinellus
Parasyrphus nigritarsis
Parasyrphus punctulatus
Parasyrphus vittiger
Parhelophilus consimilis
Parhelophilus frutetorum
Parhelophilus versicolor
Pelecocera tricincta
Pipiza austriaca
Pipiza bimaculata
Pipiza fenestrata
Pipiza lugubris
Pipiza luteitarsis
Pipiza noctiluca
Pipizella viduata
Pipizella virens
Platycheirus albimanus
Platycheirus ambiguus
Platycheirus angustatus
Platycheirus discimanus

Meligramma euchromum
Meligramma guttatum
Meligramma trianguliferum
Platycheirus clypeatus
Platycheirus occultus
Platycheirus peltatus
Platycheirus scutatus
Platycheirus splendidus
Sphaerophoria interrupta
Sphegina sibirica
Cheilosia caerulescens

Platycheirus fulviventris
Platycheirus granditarsus
Platycheirus immarginatus
Platycheirus manicatus
Platycheirus perpallidus
Platycheirus podagratus
Platycheirus rosarum
Platycheirus scambus
Platycheirus sticticus
Platycheirus tarsalis
Pocota personata
Portevinia maculata
Psilota anthracina
Rhingia campestris
Rhingia rostrata
Riponnensia splendens
Scaeva pyrastri
Scaeva selenitica
Sericomyia lappona
Sericomyia silentis
Sphaerophoria batava
Sphaerophoria fatarum
Sphaerophoria philanthus
Sphaerophoria rueppellii
Sphaerophoria scripta
Sphaerophoria taeniata
Sphaerophoria virgata
Sphegina clunipes
Sphegina elegans
Sphegina verecunda
Syritta pipiens
Syrphus ribesii
Syrphus torvus
Syrphus vitripennis
Trichopsomyia flavitarsis
Triglyphus primus
Tropidia scita
Volucella bombylans
Volucella inanis
Volucella inflata
Volucella pellucens
Volucella zonaria
Xanthandrus comtus
Xanthogramma citrofasciatum
Xanthogramma pedissequum
Xylota abiens
Xylota florum
Xylota jakutorum
Xylota segnis
Xylota sylvarum
Xylota tarda

Xylota xanthocnema
Baccha elongata
Neoascia meticulosa
Chalcosyrphus eunotus
Cheilosia urbana
Eristalis nemorum
Callicera aurata
Cheilosia albitarsis
Cheilosia lasiopa
Dasysyrphus pinastri
Epistrophe melanostoma
Eriozona erratica
Heringia latitarsis
Heringia pubescens
Heringia vitripennis