Common carder bee hovering
Photo by Alan Titcombe

# MANAGEMENT AND DRIVERS OF CHANGE OF POLLINATING INSECTS AND POLLINATION SERVICES

National Pollinator Strategy:
for bees and other pollinators in England

Evidence statements
and Summary of Evidence

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# Introduction

These Evidence Statements provide up-to-date information on what is known (and not known) about the status, values, drivers of change, and responses to management of UK insect pollinators (as was September 2018). This document has been produced to inform the development of England pollinator policy, and provide insight into the evidence that underpins policy decision-making. This document sits alongside a more detailed Summary of Evidence (Annex I) document written by pollinator experts.  For information on the development of the statements, and confidence ratings assigned to them, please see section 'Generation of the statements' below. Citations for these statements are contained in the Summary of Evidence document.

# Contents

# Main messages

## I. Status and trends of pollinators and pollinator-dependent plants

a) Past declines (1950s-1990) in species distributions of wild bees and hoverflies in the UK have slowed, but not ceased, since the 1990s, with the overall trend across species remaining downward (*established but incomplete*). Fluctuations notwithstanding, in the short-term (since 2010) trends in a number of species in these groups in the UK have stabilised (*established but incomplete*).

b) The number of registered honey bee colonies in the UK has increased over the last decade (*established but incomplete*), however we have no reliable data source for the number of managed bumble bee colonies or other managed pollinators. Although honey bees are major pollinators of several UK crops, UK stocks of honey bees are far below the levels required to act as the sole providers of pollination services to all crops. The majority of evidence indicates that wild bees are the main pollinators of UK crops (*established but incomplete*).

c) Historical declines in pollinator food (nectar) resources across Great Britain (GB) have slowed since the 1970s (*established but incomplete*), and the total potential nectar resource (estimated from 260 common nectar-producing plants) increased by 25% from 1998 to 2007, though not returning to estimated 1930 levels (*established but incomplete*). However, the diversity of nectar-producing plants has continued to decline (*established but incomplete*).

d) The service of pollination, provided by wild and managed insects, is dependent on insect numbers, and can be improved by diverse pollinator communities (*established but incomplete*). With the long term downward trend in species distributions it is possible that pollination services to crops and wild plants have also declined in the long term (*inconclusive*). Some pollination deficits (i.e. shortfalls in production due to lower than ideal levels of pollination) have been observed in UK apples, however the extent of deficits in other crops are unknown.

e) DNA methods such as illumina barcoding can be used to identify almost all UK bee species and can provide abundance data. However, in its current state of development this method may be prohibitively expensive for UK-scale monitoring, as specimens are analysed individually, increasing time and cost (*established but incomplete*).

## II. Values of Pollinators

a) Pollinators add substantial economic value to crop production through improving crop quality and quantity (*well established*). The economic benefits of pollination to crop production in the UK is approximately half a billion GBP a year, based on yield (*established but incomplete*). This estimate only considers direct benefits to current producer profit, and does not include additional economic benefits such as reduced prices for consumers, costs avoided, and natural capital benefits/losses.

b) Estimating the costs and benefits (economic and socio-cultural) of managing land for pollinators is hindered by several knowledge gaps, such as the dependence of crop varieties on pollination, the socio-cultural values of pollinators in the UK, such as landscape aesthetics and the role of pollinator diversity in these services.

## III. Drivers of pollinator status & current and future risks

### Land use

a) Habitat loss and fragmentation since the 1930s, and intensive land management practices, have negatively affected wild pollinators by reducing food (*well established*) and nesting resources (*established but incomplete*). This can lead to simplified pollinator communities dominated by common, generalist species (*well established*).

b) Chemicals used across agricultural, rural, suburban, and urban landscapes to control pests and weeds have a range of unintended direct and indirect effects on both wild and managed pollinators (*well established*).

c) Following the 2013 EU moratorium and 2018 ban, neonicotinoids still persist in soil leading to uptake and exposure to pollinators through wild plants, and in plants sold for garden use (*established but incomplete*). Explicit links to bee health, through this mode of exposure, have not been shown – however, neonicotinoid treated crops are known to have negative sub-lethal effects on bumble bees and solitary bees (*established but incomplete*) and honey bees (*unresolved*). The impact of neonicotinoid exposure on bee populations remains unclear (*unresolved*).

d) Urban insect pollinator communities are dominated by common, generalist species (*well established*). Bumble bees appear less adversely affected by urban development than other pollinator taxa like hoverflies (e*stablished but incomplete).*

## Climate change

e) Climate change has already altered the range (range contraction/poleward movement/higher elevations), abundance, and seasonal activity (overwintering behaviour) of some pollinator species, and will continue to do so; and range alteration is likely to increase in the coming decades (*established but incomplete*). The combination of climate change with other global change pressures (e.g. land-use changes, invasive alien/non-native species) are likely to pose significant future threats to pollinator communities (*established but incomplete*).

f) Currently in the UK the impact of invasive alien plant and predator species on pollinators and pollination is considered to be less profound than other pressures (e.g. land management, pesticides, pests and pathogens) (*established but incomplete*). However their importance as a driver of pollinator status may rise with climate change creating new opportunities for invasive alien species, alongside more confounding pressures from climatic or other stressors (*established but incomplete*).

## Pests & diseases

g) A major threat to honey bee populations is the ubiquitous mite *Varroa destructor* and the many bee viruses it transmits, such as the *Deformed wing virus* (DWV) (*well established*). A new, more virulent strain of DWV is currently spreading throughout honey bee populations in Europe and the UK (*established but incomplete*). Hive loss has been lessened and prevented by appropriate management of apiaries (*well established*).

h) Screening approaches in commercial bumble bee production facilities are important for reducing or eliminated the risk of transfer of pests between managed and wild bumble bee populations (*established but incomplete*).

# IV. Effectiveness of management and policy for pollinators

## Agricultural landscapes

a) Drivers of pollinator decline related to land use and land management can be mitigated by providing diverse and high-quality floral resources (*well established*); maintaining and restoring ecological infrastructure such as flowering hedgerows (*established but incomplete*); adopting 'ecologically-intensive' land management practices, including organic farming (*well established)*, and managing for ecosystem services (*established but incomplete*).

b) Empirical and modelling evidence indicate that a minimum of four years of monitoring is required to measure wild bee population responses to changes in agricultural land management (e*stablished but incomplete,* 4.1.2)

c) Sown wildflower field margins can substantially increase the plant species richness and provision of pollen and nectar resources on arable and horticultural land (*well established*). They are credited with enhancing local pollinator communities (*well established*), and reducing the local loss of bee species (*unresolved*). However their overall effect on UK pollinator populations is unknown.

d) The inclusion of particular plant species to sown wildflower strips would enhance their support of pollinators with more specialist diets (*established but incomplete*).

e) Protection and restoration of flower rich semi-natural habitats (e.g. grasslands, heathlands and broadleaved woodland) will be more effective for improving the status of pollinators if targeted to landscapes with between 1 and 10% pre-existing flower-rich semi-natural habitat (*established but incomplete*).

i) Habitat restoration of heathland, scrub, semi-natural grassland or woodland can have overall positive effects on wild bee numbers and number of species, even if activities are not specifically focused on pollinators (*established but incomplete*). Greater landscape heterogeneity may buffer long-term declines in wild bee richness at local site levels (*established but incomplete*).

f) Changes to management of existing hedgerow, field margin, improved grassland, road verges, public green spaces, and residential gardens, to optimise cutting and reduce fertilizer and pesticide use, can allow existing flowering species, which support a wide range of wild pollinators, to thrive (*established but incomplete*).

g) Nest boxes for cavity nesting bees have been successful in increasing numbers of solitary bees in their locality, over time, particularly where nests are provided in the same place in subsequent years (*established but incomplete*). Bumble bee nest boxes have not been effective in the UK (*well established*). We do not have information on the success of habitat creation for ground-nesting solitary bees and wasps.

## Urban and transport infrastructure

h) Mowing frequency and timing in the season of public greenspace, road verges, and residential gardens, influence the floral resources available for pollinators (*well established*). The optimal viable mowing regime is different for these different land uses and may be different for urban and rural road verges (*established but incomplete*): Floral resources in parks benefit the most from no mowing at all (*inconclusive*); Similarly, floral resources in gardens benefit from reduced mowing frequency (*inconclusive).*

i) Railway embankments can be managed to provide valuable refuges for floral resources and pollinators (*established but incomplete*), however the best management regime is not known. We also lack information on the effectiveness of different management approaches of brown field sites, industrial areas, airports, and university and college campuses.

j) In the urban environment, both perennial flower plantings (also known as 'meadows') comprised of perennial native species, and (to a lesser extent) those composed of annual and primarily non-native species, significantly increase pollen and nectar content compared to amenity grassland, and attract more pollinators (*established but incomplete*).

## Beekeeping

k) The key factors driving honey bee health are beekeeper experience and apicultural practices (*established but incomplete*), however we still have an incomplete picture of hive numbers, health, and management practises (*well established*).

# Generation of the statements

The Summary of evidence (Annex 1) was authored as an expert elicitation paper, and is not a systematic review of the literature. The approach to this evidence update consisted of a rapid expert assessment of the recent and main evidence about the threats and opportunities for pollinators from several policy-relevant drivers. It was authored by 10 academic experts, and was written to present up-to-date evidence relevant to informing implementation of England's National Pollinator Strategy. Some of the following topics have been reviewed previously, and in such cases the review is referred to, and additional evidence cited. Topics not previously reviewed have been covered more extensively, and include more references to original research articles. This work builds mainly from the Vanbergen at el (2014) report – 'Status and value of pollinators and pollination services'. The Summary of evidence paper was peer-reviewed by academics, and Defra group policy teams, prior to the drafting of evidence statements.

The following evidence statements were drawn from the Summary of evidence document and assigned confidence ratings, by representatives of Defra's wildlife evidence team, and the panel of authors. This set of statements and summaries were peer-reviewed by members of Defra's Pollinator Advisory Steering Group, Expert Committee on Pesticides, Nature Strategy team, and Plant Health team, and JNCC's Terrestrial Evidence team.

Each statement has been assigned a confidence level by the panel of authors, using the IPBES 4-box uncertainty model (see figure 1). For each statement, the evidence base is summarised in Annex I: summary of evidence document.

Sections are consistently numbered between the statements and summaries for easy referral to the underlying evidence. Where this deviates, corresponding section numbers are stated in brackets.
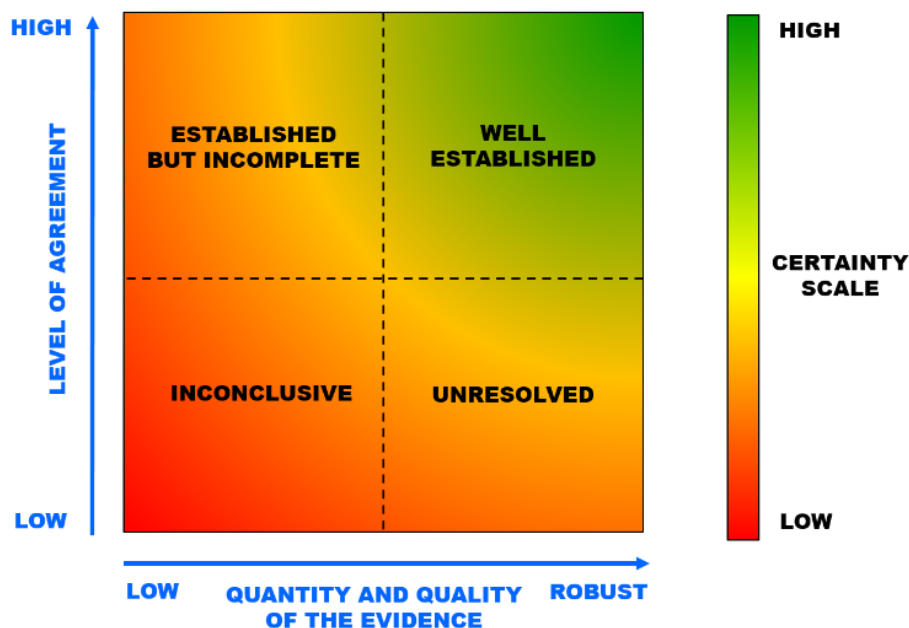


*Figure 1. IPBES 4-box uncertainty model for the qualitative communication of confidence, IPBES, Rome, 27-31 July 2015.*

# 1. Status and trends of pollinators and pollinator-dependent plants

## 1.1 Pollinators & pollinator dependent wild flowers

### 1.1.1. Wild Bees and hoverflies

- Of the 137 UK wild bee species analysed, between 1980 and 2016, 37% declined in the areas over which they were found (measured as occupancy of 1km squares), and 20% increased. Declines were sharpest between 2006 and 2013, but the average trend across species has since stabilised (*established but incomplete*).

- UK bumble bees underwent range contraction between 1960 and 1980, with rarer species having more severe declines. In recent decades the ranges of common bumble bees have stabilised, however rarer species have contracted towards their predicted climatic optimum, or retracted towards the coast (*established but incomplete*).

- Since 2010, data for bumble bee abundance has been collected, showing 9 species and two groups of species (aggregates) to be increasing (including 1 cuckoo species), and 11 species and one aggregate species to be decreasing (*established but incomplete*).

- Of the 214 UK hoverfly species analysed between 1980 and 2013, 33% of species declined in occupancy of 1km squares, and 10% increased. Their average trend gradually declined between 1987 and 2001, but has since stabilised (*established but incomplete*).

- Assessment of threat of extinction is currently underway for all the known GB bees, wasps and ants. According to the European level IUCN Red List assessment (2014) 9.2% of European bee species are considered threatened with extinction, and a further 5.2% are Near Threatened. However 57% (1101) of the species were data deficient, and so not assessed. Other national Red Lists in Europe indicate that more than 40% of bee species are threatened (*well established*).

- Efforts are being made to confirm an established population of the Short-haired bumble bee, after re-introduction attempts between 2012 and 2016.

### 1.1.2 Butterflies and moths

- Most butterflies (76% of UK species), including both specialists and generalists, have decreased in numbers or range (or both) since 1976 (*well established*).

- In the short-term (2005 to 2014), declines of several threatened butterfly species have been halted, and some species have become more abundant and widespread (*established but incomplete*).

- The major life history differences between butterflies and moths and other pollinator taxa may make them a weak proxy for bees and hoverflies (*established but incomplete*).

- According to the European level IUCN Red List assessment (2014), at the European level, 9% of butterfly species are considered threatened with extinction. Only 1% of species were classified as Data Deficient (*well established*).

- The reintroduction and restoration of the large blue butterfly to southern England in 1984 has been successful (*established but incomplete*). The reintroduction of Chequered Skipper is now underway.

### 1.1.3 All insect groups combined

- Between 1970 and 2009, an analysis of trends in distribution for 720 species of wild bees, hoverfly, butterfly, and moth across GB found that 27% of these pollinating insect species became less widespread, and 23% more widespread (*established but incomplete*).

### 1.1.4 Knowledge gaps

- Data to estimate changes in population size of bees or hoverflies at regional or national levels. Although monitoring schemes covering bumble bees and other wild pollinators are being established across the UK.

## 1.2 Managed pollinators

### 1.2.1 Honey bees

- There was a steady increase in the number of registered honey bee colonies from approximately 90,000 in 2008, to approx. 140,000 in 2012 in the UK; and in Scotland and England, the number of registered overwintering hives is continuing to increase (*established but incomplete*). As registration of honey bee hives is voluntary, this may not reflect actual numbers.

### 1.2.2 Managed Bumble bees and other managed pollinators

- The native bumble bee sub-species *B. terrestris audax* is widely used in glasshouses and open fields (*well established*) but no data on colony numbers is available as no licence is required. Managed *B. terrestris audax* colonies may pose a potential unknown risk to the wild native *B. terrestris*, and other wild pollinators (*inconclusive*).

- An estimated 40,000 to 50,000 colonies of the non-native subspecies (*B. t. dalmatinus* or *terrestris*) were imported annually to the UK in 2012 under importation licence (*well established*). However this licence terminated in 2014, and was replaced by licenses for emergency and research purposes in 2016, but only 8 have been issued to date (*well established*).

- The solitary trap-nesting bee *Osmia bicornis*, and common green bottle fly *Lucilia sericata* are currently used on a relatively minor scale in the UK (*established but incomplete*), but no data on colony numbers is available.

### 1.2.3 Knowledge gaps

- A reliable data source for the number of honey bee colonies in the UK, and colony numbers of managed *B. terrestris audax*. Likewise, data for number of *Osmia bicornis* or *Lucilia sericata* colonies currently being used.

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## 1.3 Wild plants

### 1.3.1 Wild plants

- Since the 1930s, insect pollinated wild plants have declined significantly more than non-insect pollinated wild plants, with declines being most pronounced in plant species dependent on more specialised species of pollinator (e.g. long-tongued bumblebees) (*well established*).

- Nectar resources declined in England and Wales between the 1930s and 1970s by 32%, but then stabilised by 1978. From 1998 to 2007 nectar resources increased by 25% across Great Britain, not yet returning to the 1930s levels (*established but incomplete*).

- Nectar resources have reduced in diversity up to and beyond 1990 (*established but incomplete*).

- Plant record data is rapidly accumulating for the Botanical Society of Britain & Ireland's "Atlas 2020", to be published in 2022, and under the National Plant Monitoring Scheme.

### 1.3.2 Knowledge gaps

- The role of different insect species as pollinators of wild plants.
- The impact of insect community composition and overall abundance on the reproductive success and population sizes and/or distributions of insect-pollinated plants.

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## 1.4 Pollination services and pollinator dependent crops

### 1.4.1 Pollination services

- Most crops rely on common widespread species for pollination, but different insect communities are important for pollinating different crops (*well established*).

- More diverse pollinator communities provide improved pollination service to crops and wild plants (*established but incomplete*).

- Pollination is dependent on pollinator abundance (*established but incomplete*), but as we do not have regional or national scale pollinator abundance data, estimates of change in pollination service are limited.

- The level of a crop's dependence on insect pollinators can depend on the crop variety, environmental context (e.g. soil fertility or climate), and other variables (*well established*).

- Sub-optimal pollination has been identified in some UK apple varieties (about 20% deficit) (*established but incomplete*), but the extent of these shortfalls, or incidence of excessive pollination, are unknown for most UK crops.

- Future pollination service to crops will be vulnerable where climate change creates a mismatch between optimal crop growth area, and pollinator distributions (*established but incomplete*).

### 1.4.2 Pollinator dependent crops and pollination capacity of honey bees

- While honeybees can be major pollinators of some crops, UK populations are far below levels required to meet service demands alone, reducing the UK's resilience to wild pollinator declines (*established but incomplete*).

### `1.4.3 Knowledge Gaps

- Trends in pollination service, and the abundances of key wild insect crop pollinators.

- The geographic extent and temporal changes of pollination deficits in most crops and wild flowering plants.

- Understanding how farmers use wild and managed pollinators.

- The level of insect pollination dependency for many crop varieties.

## 1.5 Potential of DNA barcoding and other techniques for pollinator monitoring

### 1.5.1 Molecular methods

- The most common next generation DNA method, metabarcoding, cannot provide abundance data, and so could only have a complementary role in pollinator monitoring (*well established*, 1.5.2).

- Illumina barcoding is known to identify almost all UK bee species and can provide abundance data, however specimens are analysed separately, increasing time and cost. In its current state of development this method may be prohibitively expensive for UK scale monitoring (*established but incomplete*, 1.5.2).

- Mitochondrial metagenomics may be used to analyse bulk samples, but would need to be further developed to attain abundance data (*well established*, 1.5.2).

### 1.5.2 Reference databases

- Using any DNA method for monitoring requires the creation and maintenance of sequence reference databases (*well established*, 1.5.3).

- A reference database exists for UK bee species, of the mitochondrial gene CO1 (cytochrome oxidase I), which can be used to identify 90% of UK bee species (*established but incomplete*, 1.5.3).

- For UK insect pollinators other than bees, a reference database would need to be created, some of which may be possible through data-mining online sources to reduce cost (*established but incomplete*, 1.5.3).

### 1.5.3 Monitoring of networks

- With further development, DNA methods could be used to study dynamics between pollinators and parasites, however this is not likely to be applied to large-scale monitoring (*established but incomplete*, 1.5.5).

- DNA methods have the potential for detection of floral visitation from samples of pollen sacks and honey (*well established*, 1.5.5).

### 1.5.4 Knowledge gaps

- An up-to-date cost analysis, to judge if new molecular/DNA methods would make monitoring more cost effective (1.5.6).

## 2.  Values of pollinators

### 2.1 Economic impacts of pollination services in the UK

- The economic value of pollination to crop production in the UK is estimated to be approximately half a billion GBP a year (*established but incomplete*, 2.1.2). However these estimates are based on generalizations of global literature, do not include the benefits to consumers, and are likely to be underestimates (*established but incomplete*, 2.1.4).

- Pollination directly increases the quantity and quality of yield in many crops (*well established*, 2.1.2). An example from the UK estimated that pollination service in grown apples increased net profit by between 48% and 68% (*established but incomplete*).

- Global evidence demonstrates that pollination can increase net profit for farmers, increase the natural capital of agricultural systems, and reduce the costs for consumers (*established but incomplete*, 2.1.2).

- Wild pollination services can save farmers the additional cost of paid pollination services, or the potential cost of mechanical pollination (*established but incomplete*, 2.1.2).

### 2.1.2 Knowledge gaps

- Level of dependence of contemporary varieties of crops on pollination for optimum output.

- How the economic benefits of pollination vary between different pollinator communities.

- The distribution of pollinator natural capital, the populations of pollinating insects that provide services to crops, across the UK - its local value, and areas of possible pollination deficit.

- The economic costs and benefits of different management options for pollinators and how can they be made more cost-effective.
- Level of resilience of UK food systems to pollinator changes, nationally and abroad.
- If pollination increase crop storage-life, reducing waste. If pollination result in greater resilience of on-farm production.

### 2.2 Socio-cultural values of pollinators and pollination

- Globally, pollinators and animal pollinated plants contribute to a wide range of socio-cultural values including aesthetic value, cultural symbolism, existence value, health values, and undiscovered economic and non-economic value (*well established*). Evidence for these values in the UK is available but highly fragmented.

- Global evidence indicates that pollination is important in underpinning the supply of micronutrients that are necessary for good quality of life (*unresolved*).

- In England the public more readily adopt the idea of pollinating insects as members of an interconnected system within the environment, rather than as providers of a service to humans (*inconclusive*).

### 2.2.1 Knowledge gaps

- The pollinators and pollinator dependent plants in the UK that have cultural value.
- How much of the UK's flora is dependent upon insect pollination.
- Level of importance of pollinated crops for health and wellbeing in the UK.
- The extent to which pollinators improve public understanding of ecology.

# 3. Drivers of pollinator status & current and future risks

## 3.1 Land use

### 3.1.1. Land-use changes in habitat and landscape structure

- Land use changes often destroy, fragment or degrade semi-natural habitat and simplify the landscape. This reduces or modifies the floral or nesting resources for insect pollinators and has been an important driver of pollinator population declines, species extinctions and changes in plant-pollinator communities and interactions, since the 1930s (*well established*).

- There is evidence (from GB) that landscape heterogeneity may buffer long-term declines in wild bee species richness at local site levels (*established but incomplete*).

- Pollinator species with more specialised requirements for food or habitat are more susceptible to land use change and land management intensification (*established but incomplete*). International studies show that abundant generalist bee species are able to persist in landscapes dominated by intensive agriculture (*well established*).

### 3.1.2. Urbanization

- Urban land cover is increasing globally, and in the UK, (*well established*). This destroys, fragments and modifies habitats (*well established*). The impact on pollinators depends on the type of land converted and the quality of urban greenspace for pollinators (*established but incomplete*)

- Towns and cities in England support higher species richness of bees compared to agricultural land although there is no difference in bee abundance between urban and agricultural areas or nature reserves (e*stablished but incomplete*).

- Agricultural areas support higher hoverfly abundances than urban areas, although there is no difference in species richness (*established but incomplete*).

- Urban insect pollinator communities are dominated by common, generalist species (*well established*). In particular, bumble bees appear less adversely affected by urban development than other pollinator taxa such as hoverflies (e*stablished but incomplete*).

### 3.1.3 Conventional intensive agriculture

- Since the 1930s conventional intensive agriculture (industrial management of large-scale monocultures with high levels of agrichemical inputs, tillage, grazing, mowing and mechanization) has reduced wild pollinator food and nesting resources, and has reduced wild pollinator diversity and abundance (*well established*). Such removal of species from agricultural pollinator communities presents a risk to the sustained and effective pollination function as the environment changes (*established but incomplete*).

- Monocultures of mass flowering crops present transient food sources for wild pollinators and do not compensate for the losses of wildflowers that sustain most pollinators through their life cycle (*well established*).

- Large-scale cultivation of mass flowering crops like oilseed rape may disrupt pollination services in the wider landscape (*established but incomplete*).

### 3.1.4 Agricultural genetically modified organisms (GMO)

- Insect-resistant (IR) GMO crops are engineered to target specific crop pests (e.g. butterfly/moth larvae, beetles). There have been reports of lethal effects of IR crops on butterfly species, but not bees or hoverflies (*established but incomplete*). Currently, few studies worldwide have examined sub-lethal effects of IR-crops on pollinator species and these provide limited and conflicting evidence of an impact (*inconclusive*, 3.1.4.1).

- Use of IR crops has the potential to reduce insecticide use, which can benefit non-target insects like bees (*established but incomplete*), but this benefit has not usually lasted as secondary outbreaks of the target, or non-target pests has often led to resumption of insecticide use (*established but incomplete*, 3.1.4.1).

- Management of herbicide tolerant (HR) GMO crops involves regular application of herbicides to reduce weed populations (*well established*), which reduces pollen and nectar sources for pollinators in farmed landscapes (*established but incomplete*). This is likely to reduce pollinator numbers and diversity (*established but incomplete*, 3.1.4.2).

### 3.1.5 Knowledge gaps

- Clarity in the roles of different pollinator taxa in wild plant and crop pollination.

- The level of habitat diversity, heterogeneity or configuration required for sustained delivery of pollination services to crops and/or wild plants.

- If habitat restoration in degraded urban and agricultural settings is effective for attracting and sustaining wild pollinators.

- Evidence of the sub-lethal effects of GMO's on pollinators.

- The effects of insect-resistant (IR) crop use, and potential associated reductions in pesticide use, on pollinator abundance and diversity.

- The effects of genes from herbicide resistant crops, when spread into wild plants.

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## 3.2 Pesticides, fungicides and herbicides

- Chemicals are used broadly across agricultural, rural, suburban, and urban landscapes to control pests and weeds, have a range of unintended direct and indirect effects on both wild and managed pollinators (*well established*).

### 3.2.1 Insecticides

- Neonicotinoid insecticides have negative sub-lethal effects on solitary bees and bumble bees at individual, colony and population scales (*well established*). Sub-lethal effects have been shown on individual honey bee (*well established*) but evidence is equivocal at colony scales and over long-time scales (*unresolved*). There is some experimental evidence of impacts on pollination (*inconclusive*).

- In the UK, amounts of neonicotinoid use in oilseed rape were significantly correlated with declines in the distribution of solitary bee species known to forage on oilseed rape (*established but incomplete*). Recent farm scale experiments in Europe including England have demonstrated a range of negative, non-significant, and positive impacts of neonicotinoids on solitary and bumble bee survival and performance at actual field exposure (*established but incomplete*).

- Laboratory and field studies of neonicotinoid impacts on honey bees have found negative, neutral, and positive effects (*unresolved*).

- Wild plants near neonicotinoid seed-treated crops, and plants sold for garden use, have been found to be contaminated by these insecticides (*established but incomplete*).

- In the UK neonicotinoids were still detectable in more than 20% of 109 honey samples collected the year after the EU moratorium was put in place (*established but incomplete*).

- Negative chronic and sub-lethal impacts of sulfoxaflor, a sulfoximine, on bumble bee colony growth and reproductive success have been found (*inconclusive*).

### 3.2.2 Other agrochemicals

- Pollinators are exposed to a broad range of fungicides and herbicides (*well established*).

- Herbicides reduce the availability of forage resources (*well established*), but how this relates to pollinator abundance or diversity remains unclear (*established but incomplete*).

- Fungicides are the most frequently detected and abundant biocides in wild bumble bees, and have been linked to their range contraction in the USA (*established but incomplete*).

- Fungicides are abundant in honey bee hive material. They kill the mid-gut cells of the bees, resulting in fungal disease, alter the microbiome, and increase the toxicity of neonicotinoids (*established but incomplete*).

- Adjuvants, applied together with agrochemicals, can increase their toxicity to bees (*inconclusive*).

- Pollinators are exposed to complex mixtures of multiple different pesticides (*established but incomplete for bumble bees and honey bees*), and these mixtures may have synergistic effects (*inconclusive*).

### 3.2.3 Knowledge gaps

- Understanding the individual and cumulative effect of total use of herbicides, fungicides, insecticides and adjuvants within a growing season on the health, survival and performance of pollinators and pollinator dependant wild plants.

- Changes in pollinator exposure to pesticides, fungicides and herbicides, to provide early warning of increasing risk, through monitoring.

- The effect of the EU moratorium and ban on the reduction of exposure to neonicotinoids, as would be predicted from half-life calculations, and the effect of the ban on pollinator populations.

- The relative impact of pesticides on pollinator populations and pollination services, compared to other drivers.

- An understanding of how farmers will respond, in terms of crop management and agrochemical use, to a permanent restriction of neonicotinoid use, and the likely consequences of these changes for pollinators.

- The potential impacts of novel and emerging agrochemicals on pollinators. How these can be assessed within the framework of risk assessment prior to their approval for use.

- The impact of chemicals in the urban and suburban environment (pesticides, artificial fertilisers, atmospheric pollution) on pollinator health.

## 3.3 Climate change

- Climate change has already altered the range, abundance, and seasonal activity of some pollinator species (*well established*), and is likely to continue in the coming decades (*established but incomplete*).

- Climate change has contributed to overall spring advancement, especially in the Northern Hemisphere (*established but incomplete*). Seasonal advancement and extreme climatic events, could potentially lead to mismatches in plant and pollinator life cycles, to the detriment of both plants and pollinators (*inconclusive*).

- Experimental studies show that shifting autumn/winter temperature can drastically affect the overwintering behaviour of insects, affecting survival of queens (*established but incomplete*).

- Bumble bee species in Europe and N. America have suffered from range contractions in their southern distributions but have failed to compensate by moving beyond their historical northern range limits (*established but incomplete*).

### 3.3.1 Knowledge gaps

- The impacts that climate change variables will have on insect pollinators, and pollination service, for example $CO_2$, precipitation, warming of seasons (other than spring)?

- The impacts of extreme events on pollinators.

- The impact of climate change on plant – pollinator interactions.

## 3.4 Invasive alien species

- Abundant alien flowering plant species can dominate the diet of pollinators (*well established*). There are only a few accounts in the UK, or comparable regions, where they have led to consistently lower pollinator diversity or abundance (*unresolved*). Effects of alien invasive species may be subtle, chronic (possibly undetected) or act in combination with other stressors (3.4.1).

- Invasive alien flowering plants can dominate pollinator interactions leading to a substantial modification of plant-pollinator networks (*well established*, 3.4.1).

- Invasive alien flowering plants can increase pollinator activity in a given location (*well established*). This can be detrimental to co-flowering native plants by drawing away pollinators, or beneficial by increasing pollinator activity overall (*unresolved*). The precise outcome in a given situation depends on the abundance, distribution and floral rewards of the invading plant compared to the local native species (*well established*, 3.4.1).

- The Asian hornet, if it establishes in the UK as in Western Europe, could gain high population density within 10 years, and would be an additional threat to honey bee populations (*established but incomplete*, 3.4.2).

- The Asian hornet also attacks wild pollinators and is a floral visitor so there are potential risks to the network of interactions affecting pollinator communities and pollination processes (*inconclusive*, 3.4.2).

### 3.4.1 Knowledge gaps

- Forecasts of the risks of emerging invasive species (e.g. Asian hornet) for managed and wild pollinators. At the time of writing Defra's Asian hornet contingency policy is under review.
- The consequences of species invasions (plants, animals or pathogens), for the structure and stability of ecosystems that comprise a range of functional species groups (mutualists, decomposers, herbivores etc.).

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## 3.5 Pests and diseases of bees

### 3.5.1 Impacts on honey bees

- The invasive mite *Varroa destructor* is ubiquitous across England and, combined with the co-occurring *Deformed wing virus*, poses a major threat to honey bee health. A new, more virulent strain of *Deformed wing virus*, linked to overwintering honey bee mortality, is spreading throughout honey bee populations in Europe, including the UK (*established but incomplete*).

- *Varroa destructor* and *Deformed wing virus* are managed in UK honey bee populations by bee husbandry techniques and chemical treatment. A study including England and Wales found honey bee colony loss from *V. destructor* infestation was predicted by beekeeper background and apicultural practices, indicating the importance of appropriate management of this pest (*well established*).

- European (EFB) and American (AFB) foul broods remain at low levels (less than 2% and less than 1% respectively), with no recent (2013-2018) changes in incidence in managed honey bees in the UK (*well established*).

### 3.5.2 Impacts on wild bumble bees

- Wild bumble bees share viral and fungal pathogens with managed honey bees in the UK, and these pathogens have been shown to negatively impact bumble bees in the lab. However, it is unknown if, and in which direction, these pathogens would spread in the field. It is unknown if these pathogens have any impact on wild bumble bee populations (*established but incomplete*).

- Parasites and disease can be transmitted from commercial to wild bumble bees, however new pathogen screening approaches in commercial bumble bee production facilities may have reduced or eliminated this risk (*established but incomplete*).

### 3.5.3 Impacts on wild solitary bees and other pollinators

- In a German study, viruses found in honey bees were found to infect and replicate in solitary bees (also native in England), and transmission to wild solitary bees is likely. In England, honey bee viruses were found in hover flies, but there was no evidence of viral replication (*established but incomplete*).

### 3.5.4 Knowledge gaps

- Whether pathogens and parasites are transmitted from commercial to wild bumble bees in the UK, and the impacts on wild bumble bee populations.

- The epidemiology of viruses and other pathogens in managed and wild pollinators, including directionality of transmission.

- The impacts emerging diseases might have on wild pollinator populations.

- The relative impact of pests and diseases on managed and wild pollinator populations, in comparison to, and in combination with, other drivers of population change.

- Assessment of the risk of commercial movement and use of managed pollinators – honey bees, bumble bees, solitary bees, hover flies – to act as sources for disease emergence in managed and wild pollinators.

- The effectiveness of current control and management strategies at preventing novel invasive pests establishing in England, in particular *Tropilaelaps* mites, and the small hive beetle, as these have not been intercepted in England to date. There is a need to devise effective controls where current measures are lacking.

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# 4  Effectiveness of management and policy for improving the status of pollinators

## 4.1 Rural – Agriculture & Nature Conservation

### 4.1.1 Creation of wild flower habitat

- Plant species richness on arable and horticultural land increased by more than 30% between 2000 and 2007, partly due to an increase in sown wildflower field margins which are used by wild bees (*established but incomplete*, 4.1.2).

- Flower strips, naturally regenerated uncropped margins, and extensively managed grassland (and organic farming) lead to local increases in abundance and number of species of bees, hoverflies and butterflies found in those habitats (UK and European studies) (*well established*, 4.1.2).

- Provision of floral resources enhances bumble bee colony reproduction, and the proportion of high-value foraging habitat in the colony's vicinity (up to 1,000m) increases the survival of the bumble bee population (*established but incomplete*, 4.1.2).

- Empirical and modelling evidence indicate that a minimum of four years of monitoring is required to measure wild bee population responses to changes in agricultural land management (*Established but incomplete,* 4.1.2)

- The contribution of agri-environment schemes to national nectar provision was low in 2007, with calcareous grassland, broadleaved woodland, neutral grassland and shrub heathland being the most important habitats for nectar provision. Together, white clover, ling heather, marsh thistle and bell heather were estimated to produce over 50% of the UK's nectar supply (*established but incomplete*, 4.1.2).

- Well managed nectar flower mixtures have the highest nectar productivity (kg sugars/ha/year) among five agri-environment options for which this can be compared at national scale (compared with wild bird seed mixtures, enhanced grass buffer strips, hay-making supplement and species-rich semi-natural grassland). Despite increased uptake under the Countryside Stewardship Scheme since 2015, their contribution to the national nectar resource represents less than 1% of England's nectar provision (*established but incomplete*, 4.1.2).

## 4.1.2 Factors determining success of habitat creation in improving the status of pollinators

- The factors determining the success of habitat creation at supporting pollinators are: the magnitude, type and quality of the floral resources provided, and the landscape context (*well established*, 4.1.3).

*Magnitude and quality of floral resources*

- The density of floral resources in a habitat is almost always a key predictor of both abundance and species richness of the pollinator community found foraging there (*well established*, 4.1.3.1).

- A set of farms with Higher Level Stewardship in SE England had higher floral resources than those with Entry Level Stewardship, which was associated with higher nest densities for four common bumble bee species, but overall bee diversity was not enhanced *(inconclusive*, 4.1.3.1).

- Woody plants are important sources of nectar and pollen for bumble bees and honey bees in spring and early summer (*well established*). At the end of summer, ivy was found to be a key resource (*established but incomplete*, 4.1.3.1).

- Solitary bees rely on plants not typically included in seed-mixes for agri-environment schemes, such as scentless mayweed, hogweed, and dandelion (*established but incomplete*, 4.1.3.1).

- A diversity of floral species is needed to provide adequate nutrition to bees at different life stages. The absence or presence of particular nutritional components, and their balance, can confer particular health benefits for bees, for example reduce parasite loads *(established but incomplete*, 4.1.3.1).

- Cutting and mowing frequency of agri-environment options including hedges and field margins influence floral resources available for pollinators (*established but incomplete*, 4.1.3.1).

- Farmer experience influences the quality of flower-rich habitats produced by agri-environment scheme options, with more experienced farmers producing habitats with greater numbers of flowers for pollinators (*established but incomplete*, 4.1.3.1).

*Landscape context*

- Pollinator conservation actions such as enhancing flower densities provide the greatest benefit for pollinators in simple landscapes (defined as landscapes with 1-20% semi-natural habitat) (*well established*, 4.1.3.2).

- Benefits of Agri-environment schemes for pollinators are inconsistent in intensively farmed 'cleared' landscapes (<1% semi-natural habitat) (*established but incomplete*). This is thought to be because there are few pollinating insects left, or because of a lack of nesting and over wintering sites (*inconclusive*, 4.1.3.2).

- Within intensively farmed areas, restored flowering hedgerow, and other linear landscape features, can facilitate pollinator movements and promote bee populations and diversity (*established but incomplete*, 4.1.3.2).

## 4.1.3 Provisions for nesting

- Artificially exposed areas of bare soil can be successfully colonised by ground-nesting solitary bees and wasps in the first or second year. There is an example, from the USA, of sustainably managing the ground-nesting alkali bee in large and growing nesting aggregations over decades, for its pollination services (*established but incomplete*, 4.1.4).

- Nest box provision for cavity nesting bees can increase the numbers of solitary bees over time (*established but incomplete*, 4.1.4).

- Bumble bee nest boxes are not effective in the UK. This may be because the boxes are poorly designed, in the wrong places, or there are plenty of natural nest sites (well *established*, 4.1.4).

## 4.1.4 Recent land management

- In England the addition of flower-rich field margins through agri-environment schemes is effective at enhancing local pollinator numbers. However, it is not at sufficient scale, or appropriately targeted, to reverse observed national declines in wild pollinators. This is also likely to be the case in Scotland, Wales and Northern Ireland (*established but incomplete*, 4.1.5).

- The pollinator species known to be declining are mostly habitat or diet specialists. Half of the declining English wild bee species associated with farmland are specialised in their diet. To support these species, specific flowers need to be included in sown nectar flower strips in field margins (*established but incomplete*, 4.1.5).

- Large-scale protection and restoration of flower-rich semi-natural habitats such as grasslands, heathlands and broadleaved woodland are important for improving the status of pollinators (*established but incomplete*, 4.1.5)

- Large-scale changes to management of existing hedgerow, field margin, improved grassland and road verges, to reduce cutting, fertilizer and herbicide use, would allow existing flowering species that support a wide range of wild pollinators to thrive (*established but incomplete*, 4.1.5).

- Organic farms generally have higher pollinator densities and numbers of wild pollinator species than non-organic farms (*well established*) and have been shown to have higher floral resources and plant diversity within fields (*established but incomplete*)

- Risk-based regulations in the UK decreased the overall environmental risk of pesticides for fruit and arable crops from 2002 to 2008, but this has not been analysed for the more recent time period since 2009, or following introduction of the Plant Protection Products (Sustainable Use) Regulations 2012, during which time neonicotinoids became widely used (*established but incomplete* 4.1.5.3). Further actions such as risk assessment, regulation, labelling, training and education of pesticide users, effective promotion of Integrated Pest Management (IPM), pesticide use reduction targets and bans or moratoria, may reduce the impact of pesticides on pollinators (*unresolved*, 4.1.5).

## 4.1.5 Relevance of different management actions

- Management of agriculture through ecological intensification would mitigate the drivers of pollinator decline in intensive agricultural landscapes. This involves actively managing farmland to increase the intensity of the ecological processes that support production, such as biotic pest regulation, nutrient cycling and pollination. This approach makes use of natural functions and services to enhance agricultural productivity, and reduce reliance on agrochemicals and further land-use conversion (*established but incomplete*, 4.1.6).

## 4.1.6 Rural - Nature Conservation

- Habitat restoration in non-agricultural settings such as heathland, scrub or woodland has overall positive effects on wild bee abundance and species richness, even if restoration activities are not focused on pollinators specifically. Restoration techniques usually focus on restoring the plant community, and included burning, grazing, seeding and removing invasive plants (*established but incomplete*, 4.1.7).

*Reintroduction of Bombus subterraneus*

- Efforts to reintroduce the Short-haired bumble bee (started in 2009), by the Bumblebee Conservation Trust, has so far created over 1,300 hectares of flower rich habitat. The reintroduction effort has resulted in: the reappearance of the Ruderal bumble bee (after a ten year absence at the site); the Brown-banded carder bee; the Moss carder bee; and the Red-shanked carder bee, in addition to the Short-haired bumble bee. It is not yet known if the Short-haired bumble bee has fully established on the site.

*Other initiatives*

- Many nature conservation projects create, restore and manage pollinator friendly habitats. However, there is currently no published scientific evidence of their effectivity at supporting pollinators.
- Wider nature and nature conservation projects like Coronation Meadows and Save our Magnificent Meadows create, restore and conserve, flower rich meadows.

- [B-Lines](#) is a nationwide initiative to restore and create a series of wildflower-rich habitat stepping stones aiming to link existing wildlife areas together to create network of habitats with an aim to weave them across the British landscape.
- [The Glastir Small Grants](#) is a programme of capital available to farming businesses across Wales to carry out projects that will help to improve and maintain the traditional landscape features in Wales, and provide habitat linkage for pollinating insects.
- Defra grants to Local Nature Partnerships projects in Hertfordshire, Lancashire, Lincolnshire, Durham and Surrey support schools and museums to develop pollinator friendly courses, promote ecological understanding and awareness as well as create roadside habitats for bees and butterflies.

## 4.1.7 Knowledge gaps

- Measurement of the trends in pollinator numbers, reproductive success, and survival, so the benefits of wildflower habitat creation can be determined.
- The distribution, and enhancement, of nesting resources for ground-nesting bees in UK agricultural landscapes. This information would be key to designing appropriate conservation actions to support ground-nesting bees.
- Analysis and mapping of the contribution of different habitat types and particular plant species to the provision of pollen resources, and to the specific nutritional requirements of wild pollinators, at landscape scale.
- The effectiveness of the provision of nesting resources for bees, including cavity-nesting and ground-nesting bees, in the UK.
- Population-level impacts of management actions on pollinators.
- Spatial analysis of UK farmed landscapes, to quantify the proportion of flower-rich semi-natural habitats, and identify areas with between 1 and 10% of such habitats. Identification of such areas would inform where pollinator conservation actions would be most effective.
- Quantification of the effects of Integrated Pest Management (IPM) and other changes to the pesticide regime, on pollinator diversity, abundance and health in farmland.

## 4.2 Urban and transport infrastructure

### 4.2.1 Management of urban areas for pollinators

- Floral abundance and numbers of bee visits are higher in allotments and gardens compared to other urban land uses, numbers of hoverfly visits are higher in allotments and gardens compared to road verges, pavements and other urban greenspace (*established but incomplete*). Gardens in neighbourhoods with higher household income have more pollinator visits to flowers, more flowers and higher flowering plant species richness (*established but incomplete*).

### 4.2.2 Mowing of public greenspace, road verges and residential gardens

- Mowing frequency and timing in the season of public greenspace, road verges, and residential gardens, influence the floral resources available for pollinators (*well established*). The optimal viable mowing regime is different for these different land uses and may be different for urban and rural road verges (*established but incomplete*): Floral resources in parks benefit the most from no mowing at all (*inconclusive*); Similarly, floral resources in gardens benefit from reduced mowing frequency (*inconclusive*).

### 4.2.3 Management of railway embankments

- Along railway embankment management, bare ground has a positive effect on numbers of bees and bee species, and shrub density has a negative effect (*established but incomplete*). Numbers of bee and hoverfly species are positively related to wood cover in the surrounding landscape, while the number of bee species is positively affected by the diversity of non-native flowering plants (*established but incomplete*). Numbers of butterflies, and butterfly species benefit from a diversity of native flowering plants, with bare ground having a negative impact (*established but incomplete*).

- Railway embankments can function as corridors for some pollinator taxa, and can be valuable refuge areas for floral resources and pollinators (*inconclusive*).

- Urban flower plantings (also known as 'meadows') comprised of either perennial native species, or a mix of annual native and non-native species, significantly increase pollen and nectar resources compared to amenity grassland areas (*established but incomplete*).  These plantings can attract more pollinators than control plots of amenity grassland (*established but incomplete*).

## 4.2.4 Choice of floral species for plantings

- The most beneficial plantings for pollinators are species that produce large quantities of pollen and nectar that are accessible to UK pollinators. Data on pollen and nectar are available for UK native, and some non-native plant species (*established but incomplete*).

- Plantings comprising native and near native species attract more pollinators than exotic species, although using exotic plants to extend the flowering season is beneficial for pollinators later in the year (*established but incomplete*).

- Some horticulturally modified plant varieties are less beneficial for pollinators than non-modified varieties, as the modified flowers generally contain less nectar which can be less accessible to pollinators (*established but incomplete*).

- Many lists of plants that are recommended for pollinators exist, e.g. the Royal Horticultural Society's Plants for Pollinators, but none are based on published data (*established but incomplete*).

## 4.2.5 Use of pesticides

- Neonicotinoid pesticide use in urban lawns negatively affects bumble bee colony growth, and new queen production, when applied to blooming plants (*established but incomplete*).

- There is a negative correlation between butterfly and bumble bee abundance, with insecticide and herbicide use in gardens. Their use had a larger negative effect on butterfly and bumble bee abundance in gardens in highly urbanised areas (*established but incomplete*).

- Bumble bees from ornamental urban gardens contain a wide suite of neonicotonoid insecticides, and fungicides, with concentrations and detection frequencies lower than in bees from farmland sites (*established but incomplete*).

## 4.2.6 Green roofs

- Green roofs attract native bees, albeit at lower abundance and diversity than in nearby city park green spaces and natural prairie areas. Seed set of native plant species was still high, suggesting that pollinator services are available on the green roofs (*established but incomplete*).

- 'Bee hotels' on green roofs are more successful on lower buildings and in areas with increased areas of green space (*established but incomplete*).

## 4.2.7 Nesting sites

- UK gardens have been found to contain greater density and survival of bumble bee nests, compared to agricultural and woodland habitats (*established but incomplete*).

- The location of artificial 'bee hotels' affects the species that use them. A study in Toronto found: the greatest native bee abundance in residential gardens; the greatest abundance of introduced bee species on rooftops and community gardens; and the greatest native wasp abundance in urban parks (*established but incomplete*).

## 4.2.8 Knowledge gaps

- If urban plantings increase pollinator abundance or just draw pollinators from other sites.
- The effect of planting non-native taxa on pollinators and the ecosystem more widely.
- The quantity of pollen and nectar that non-native plants provide and how important are they for pollinator diets.
- The benefits of green roofs and walls for pollinators in the UK.
- Information on nesting sites used by pollinators in urban landscapes.

- Determining the suitability of sites, and effectiveness of different management approaches, of railway embankments, brown field sites, industrial areas, airports, and university and college campuses; and to determine the effect on pollinators of different mowing regimes of urban and rural road verges.

## 4.3 Beekeeping and pollinator management

### 4.3.1 Honey bees

- The key factors driving honey bee health are beekeeper background and apicultural practices for bee nutritional health; and suppressing Varroa mites and the viruses they transmit. However, as registration with the National Bee Unit is optional we have an incomplete picture of hive numbers, health, and management practises (*well established*).

- The combined effect of current domestic and EU policy, and effort of the National Bee Unit and British Beekeeping Association in training beekeepers, surveying honey and managing disease outbreaks have stabilised outbreaks of European and American Foulbrood disease (*well established*).

- High densities of managed honey bees (and bumblebees) can have negative impacts on wild pollinators through competition, by depleting pollen and nectar resources or displacing wild pollinators, but this is not always the case (*unresolved*). There is a current scientific debate about whether actions specifically to support managed pollinators, such as promoting bee keeping and managed bee health, should or should not be included as part of wider pollinator conservation strategies.

### 4.3.2 Bumble bees

- Commercial producers of the native bumble bee subspecies *Bombus terrestris audax* practice disease screening and provide guidance on use and disposal of commercial colonies (*well established*). We do not know how effective this is at preventing transfer of diseases to wild populations of bumble bees or managed honey bees (*inconclusive*).

### 4.3.3 Solitary bees

- There is no management practice to control viruses in commercially produced solitary bees. Commercial sale, transport, and use of solitary bees is not covered by policy as honey bees and bumble bees. They are captured under the EU veterinary checks legislation but as there are no relevant conditions for health checks of solitary bees, they are dealt with on a case by case basis (*well established*). We do not know how trade in commercial solitary bees may impact wild pollinators through disease transmission (*inconclusive*).

### 4.3.4 Other pollinators

- Hoverflies and green bottle flies are produced commercially for pest control (*well established*), and their potential use as managed pollinators is growing (*established but incomplete*). There is no specific policy or practice guidance for hoverflies on importation, management for colony health or health of other managed or wild pollinators (*inconclusive*).

### 4.3.5 Knowledge gaps

- Further understanding of the numbers, distribution, and health of managed honey bee hives across England, and determining the effectiveness of current management on hive numbers, and effectiveness of different methods for the detection and treatment of diseases.

- Information and data on the import, sale, and distribution of managed solitary bees, hoverflies, and other flies used as pollinators, and develop targeted and appropriate policy.

- The impacts of trade and use of managed honey bees, bumble bees, solitary bees, and hoverflies on wild pollinator population

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# MANAGEMENT AND DRIVERS OF CHANGE OF POLLINATING INSECTS AND POLLINATION SERVICES

National Pollinator Strategy: for bees and other pollinators in England

## ANNEX I - Summary of Evidence

January 2019

## Introduction

This Summary of Evidence provides up-to-date information on what is known (and not known) about the status, values, drivers of change, and responses to management of UK insect pollinators (as was September 2018). This document has been produced to inform the development of England pollinator policy, and provide insight into the evidence that underpins policy decision-making. This document sits alongside a more condensed Evidence Statements document.

### Generation of evidence summary

This summary of evidence was authored as an expert elicitation paper, and is not a systematic review of the literature. The approach to this update consisted of a rapid expert assessment of the recent and main evidence about the threats and opportunities for pollinators from several policy-relevant drivers. It was authored by 10 academic experts, and was written to present up-to-date evidence relevant to informing England's National Pollinator Strategy. Some of the following topics have been reviewed previously, and in such cases the review is referred to, and additional evidence cited. Topics not previously reviewed have been covered more extensively, and include more references to original research articles. This work builds mainly from the Vanbergen at el (2014) report – 'Status and value of pollinators and pollination services'. This summary of evidence paper was peer-reviewed by academics, and Defra group colleagues, prior to the drafting of evidence statements.

A set of evidence statements were drawn from this summary of evidence, and assigned confidence ratings. Statements and ratings were assigned by representatives of Defra's wildlife evidence team, and the authors of the summary of evidence. This set of statements and summaries were peer-reviewed by members of the stakeholder group - Pollinator Advisory Steering Group.

# Contents

❖ Lead authors are noted for each sub-section.

# 1. Status and trends of pollinators and pollinator-dependent plants

Authors: Tom D Breeze, Claire Carvell, Mike P Garratt and Simon G Potts

The Vanbergen *et al.* (2014) report summarised evidence on the status and known trends of pollinators and pollinator-dependent plants in Britain. It concluded that there was good evidence that species diversity and distributions of wild bees and other taxa are changing, with more areas showing a loss of species than an increase. The report also highlighted the lack of long-term standardised monitoring of bees and hoverflies, important pollinators of crops and wild plants across the UK. Several new studies have added to the growing evidence for declines in many pollinating insect species (including managed honey bees) and pollinator-dependent wild plants, in the UK and across Europe, but also point to increases in range or a slowing of declines for some species and habitats; these studies are summarised below. Progress has been made towards implementing standardised monitoring since 2014. The Bumble bee Conservation Trust has reported population trends for UK bumble bees generated from standardised Bee Walks between 2010 and 2017. Furthermore, a national Pollinator Monitoring and Research Partnership between Government, academic institutions and NGOs has been formed, developing approaches through a design and testing phase (Carvell *et al.* 2016) followed by a new UK Pollinator Monitoring Scheme (PoMS) initiated in 2017, although at the time of writing it was too early to report on the Scheme's findings.

## 1.1 Wild pollinators - Claire Carvell

### 1.1.1 Wild bees and hoverflies

Records of species occurrence submitted by volunteer recorders, collated and verified by the Bees, Wasps and Ants Recording Society (BWARS) and the Hoverfly Recording Scheme (HRS), provide valuable information about long-term changes in species' distributions and status. Extracting this information and using it to analyse trends over time has proved difficult, because records are not collected in a standardised way and are subject to many forms of bias (both in time and space). However, recent analytical developments aimed at accounting for these biases while estimating robust trends have shown great promise (Van Strien *et al.* 2013, Isaac *et al.* 2014). Specifically, occupancy-detection models have made it possible to estimate reliable trends for individual species at the 1km square scale. Occupancy is measured as the proportion of occupied 1km grid cells across the UK in each year, with the models simultaneously estimating and accounting for variation in detection probability (Powney *et al.* 2017). This contrasts with previously published analyses which have tended to measure change in broader biodiversity metrics, such as species richness at 10km scales or greater (Carvalheiro *et al.* 2013; Vanbergen *et al.* 2014).

Trends in occupancy of 213 wild bee and hoverfly species between 1980 and 2010 were combined into a composite indicator of average relative change in species' distributions, to produce the first Indicator of the Status of Pollinating Insects published alongside other UK Biodiversity Indicators in 2015 (Powney *et al.* 2016). New Indicators were published in 2017 and 2018, with the latter, and most recent (at the time of writing) reporting changes in distribution of 351 species of wild pollinator across the UK (representing approximately 70% of all species) between 1980 and 2016 (Powney *et al.* 2017 and 2018). Only species for which reliable trends could be estimated were included; hence species with fewer than 50 total records across the time period were excluded from the 2018 Indicator. A lag in submission and collation of hoverfly records means that post 2013 records per year tended to drop off. Therefore, for the composite indicator and assessment, hoverfly occupancy estimates post 2013 were held at their value in 2013. Further, bee species were filtered, following expert consultation with

BWARS, so that only species considered to pollinate wild plants and/or crops were included (for example, 'cuckoo' species were excluded, see Powney *et al.* (2018) for a full list of species covered). The increase in species (since 2015) for which robust trends can be generated has been made possible by an increase in verified records and improvements to the modelling approach. Whether an individual species is increasing or decreasing is defined by its rate of annual change across the time period of interest (figure 1.1). Here we summarise these latest results.

Between 1980 and 2016, 14% of the 351 species analysed became more widespread (5% showed a strong increase at above a threshold of +2.81% per annum) and 34% of species became less widespread (13% showed a strong decrease at below a threshold of -2.73% per annum, equating to a decrease in occupancy of -50% over 25 years). As expected, species show considerable variation through time (Powney *et al.* 2018) and care is needed when interpreting average trends across species with contrasting ecological traits. When combined into a composite average trend across all species, occupancy declined by 22% between 1980 and 2016 and the pollinator indicator was therefore assessed as declining over this period. In the shorter term between 2011 and 2016, the indicator showed a minor increase of 2%, however given the uncertainty, the short-term trend was assessed as 'stable'. This is perhaps not surprising given the variation in individual species trends and the short time frame of this period, but is consistent with previous findings that declines in species richness of wild bees and hoverflies observed prior to 1990 may have slowed down in recent decades (Carvalheiro *et al.* 2013; Vanbergen *et al.* 2014). The analysis by Ollerton *et al.* (2014) suggested that the cluster of pollinator extinctions that occurred in 1988-1990 pointed to a future faster rate of extinction. However no species appear to have been lost subsequently, reducing the observed rate of extinction for the period 1986-1997, further suggesting that the rate of loss has slowed down (Ollerton pers. com. 2018).

The Indicator is presented separately for bees and hoverflies. Of the 137 wild bee species analysed, a greater proportion of species were declining than increasing, 37% and 20% respectively between 1980 and 2016 (figure 1.1). Averaged across bee species, the indicator shows a particularly sharp decline in occupancy between 2006 and 2013, after which it stabilised, although the bee index in 2016 was estimated 17% lower than in 1980. In contrast, the hoverfly index (figure 1.2) shows a gradual decline from 1987 to 2001, after which it remains relatively stable. Similarly to the bee results, of the 214 UK hoverfly species analysed, a greater proportion of species were declining than increasing, 33% and 10% respectively, between 1980 and 2013 (figure 1.2).

As individual species become more or less widespread, their relative abundances are likely to change and the communities in a given area become more or less diverse, though this has not been assessed. This has implications for pollination service since more diverse pollinator communities are usually more effective at pollinating a range of crops and wild plants (reviewed in Ollerton, 2017)
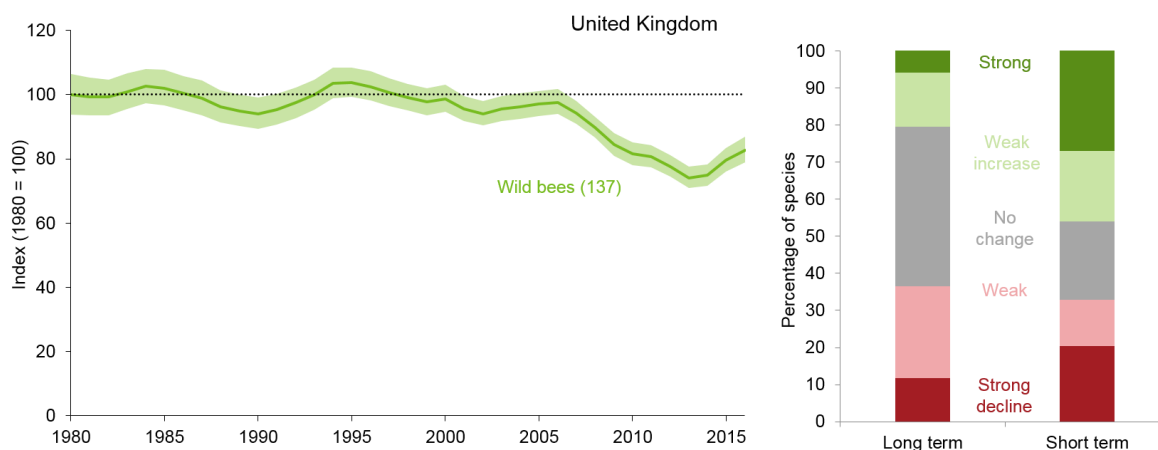
**Figure 1.1:** Change in the distribution (occupancy index within 1km squares) of pollinating wild bee species (n = 137) in the UK between 1980 and 2016. The shaded region is the 90% credible intervals of the annual occupancy estimates and represents the uncertainty surrounding the annual estimates. 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 (1980-2016) and b) the short-term (2011-2016). Thresholds used to define species trends were: Strong increase, above +2.81% per annum; Weak increase, between +1.16% and +2.81% per annum; no change, -1.14% to +1.16% per annum; Weak decrease, between -2.73% and -1.14%; Strong decrease, below -2.73% per annum (JNCC, 2018). With permission from Powney *et al.* (2018).
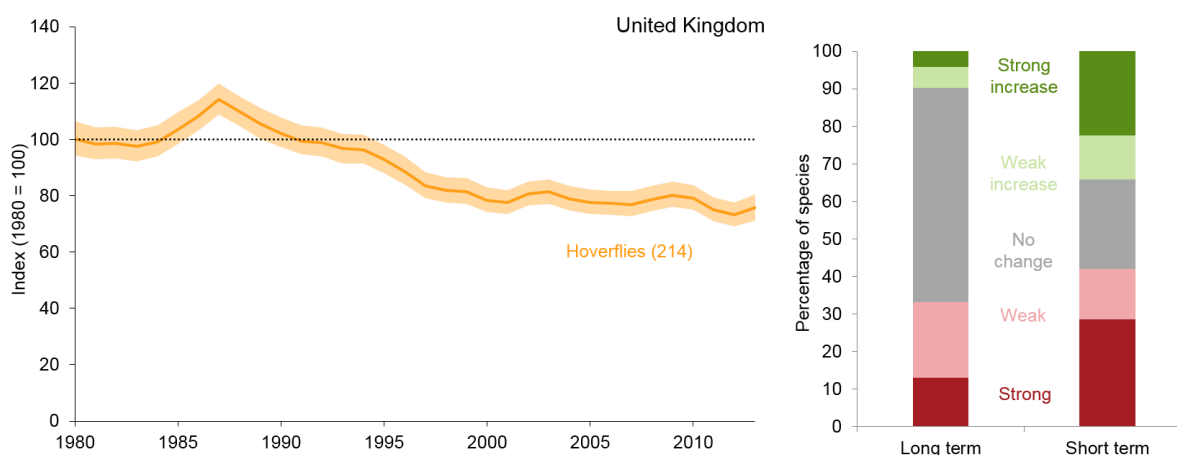


**Figure 1.2:** Change in the distribution of hoverfly species (n = 214) in the UK between 1980 and 2013. The shaded region is the 90% credible intervals of the annual occupancy estimates and represents the uncertainty surrounding the annual estimates. The solid line illustrates the rescaled indicator 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 (1980-2013) and b) the short-term (2008-2013). Thresholds given in Fig. 1.1 (JNCC, 2018). With permission from Powney *et al.* (2018).

A complete review of all the taxa within the known GB aculeate Hymenoptera (including bees) according to IUCN Red List Guidance, is currently underway and will provide a comprehensive update on any changes in extinction risk to threatened bee species (Edwards *et al.* pers. comm.). This requires the classification of individual taxa, based on set criteria, with the emphasis on demonstrable changes in abundance or distribution over the assessment period, plus a consideration of very restricted distributions. It is not directly comparable with earlier published UK Red Lists which were based on counts of occupied hectads only. The review used, for the first time for any invertebrate group, the occupancy-detection models of species trends reported in the indicator (Powney *et al.* 2017) in combination with other assessments of change in range and expert opinions.

At the European level, a review of the status of European bee species according to IUCN regional Red List assessment guidelines was published in 2014 (Nieto *et al.* 2014). This indicates that 9.2% of bee species are considered threatened with extinction and a further 5.2% are Near Threatened. However, for 1,101 (57%) of the 1,965 species assessed, there was not enough information to evaluate their risk of extinction and these species were classified as Data Deficient (Nieto *et al.* 2014); other national Red Lists in Europe indicate more than 40% of bee species are threatened (IPBES, 2016).

As bumble bees are better studied across Europe than many other wild bee species, Nieto *et al.* (2014) were able to report on the status and trends of the 68 *Bombus* species present across Europe. Of these, 23.6% are threatened with extinction and 45.6% of bumble bee species were found to have a declining population trend at European level. Only 8.8% of species were classed as Data Deficient (Nieto *et al.* 2014). In another study using distributional records of bumble bees from the UK and Ireland, Casey *et al.* (2015) found that most of the rarer bumble bee species showed marked range contractions between 1960 – 1980, with their ranges remaining more stable between 1981 – 2012. The more common species' ranges contracted, but less severely than the rare species, between 1960 and 1980, and they appear to have expanded in range post-1980. However, rare species tend to have become more marginal in the sites they occupy post-1980, some contracted towards their predicted climatic optimum and some also retracted towards coastal areas. The analyses in Casey *et al.* (2015) did not fully account for patterns of recorder effort which may distort real trends, highlighting again the need for long-term monitoring of pollinator abundance.

The Bumble bee Conservation Trust (BBCT) has reported population trends for 20 UK bumble bee species and 3 species aggregates generated from standardised Bee Walks (repeated monthly transect counts giving estimates of abundance from March to October) between 2010-2017 (Comont & Dickinson, 2018). Population trends were positive for nine species and two species aggregates. This included total bumble bees, one cuckoo bumble bee (*B. vestalis*) and four conservation-priority species (*B. ruderarius, B. humilis, B. monticola* and *B. sylvarum*). Conversely, population trends were negative for 11 species and one aggregate species. This includes the remaining five cuckoo species and two conservation-priority species, *B. muscorum* and *B. ruderatus*. Unsurprisingly the range-expanding Tree bumble bee *B. hypnorum* (which arrived in the UK around 2000) continues to increase in local abundance on BeeWalks. As with the trends in occupancy reported above, individual bumble bee species showed considerable variability in abundance over time and across the 400+ transects walked across GB (Comont & Dickinson, 2018).

Another bumble bee species, *Bombus subterraneus* (the Short-haired bumble bee), has been the focus of a significant reintroduction effort led by BBCT on the RSPB Dungeness reserve in Kent. After five years of reintroductions from Swedish populations between 2012 and 2016, coupled with habitat restoration work, the project is now focussing efforts on trying to find an established population (Gammans, 2017). DNA collected from putative workers will be barcoded to confirm they are indeed *B. subterraneus*, alongside further surveys, and thus provide evidence of establishment.

In summary, while the large-scale and long-term distribution datasets offer unparalleled opportunities for tracking large-scale changes in species distributions, they provide no information on abundance and hence population size. Despite improvements in the occupancy models described above since 2014, these remain temporally and spatially biased. Aside from the Bumble bee Conservation Trust's network of 'BeeWalk' transects running since 2010, there are currently no equivalent standardised data on changes in abundance of wild bees or hoverflies for any country across the globe.

## 1.1.2 Butterflies and moths

Trends in the abundance of butterflies and moths may provide an indicator of the level of threat to flower-visiting insects generally. While our understanding of the functional role of Lepidoptera as pollinators (in particular for crops) remains limited, the major life history differences between butterflies and moths and other pollinator taxa, may make them a weak proxy for bees and hoverflies. Nevertheless, long-term standardised monitoring of individual butterfly and moth species has allowed repeated updates of their status and trends since around 1976 (Fox *et al.* 2015). For butterflies, new analyses since Vanbergen *et al.* (2014) confirm that both habitat specialists and wider countryside species (in total 76% of all UK species) have decreased significantly in abundance or occurrence (or both) since 1976. However, as with some bee species, the short-term picture between 2005 and 2014 suggests that declines of several threatened butterfly species have been halted and a range of other species have become more abundant and widespread (Fox *et al.* 2015). The restoration and reintroduction of the Large Blue Butterfly to southern England in 1984 has been deemed a success, and a further reintroduction project is underway to restore populations of Chequered Skipper, both testament to the successful local and regional conservation of priority habitats for these species (see Section 4.1.7).

Recent analyses have also compared population estimates for widespread UK butterfly species derived from the citizen science Big Butterfly Count (BBC) project with those from the long-running standardised data cited above (Dennis *et al.* 2017). These found correlations between the two schemes, suggesting that the simple sampling protocol used to collect BBC data produced comparable estimates of butterfly abundance to more intensive methods, at least for the more common species.

At the European level, 9% of butterfly species are considered threatened with extinction according to IUCN regional Red List guidelines (a similar figure to that for bees) (van Sway *et al.* 2010). However, since butterflies are extensively recorded and monitored across Europe, only 1% of species were classified as Data Deficient.

## 1.1.3 All insect groups combined

Two key studies have provided new evidence (since 2014) for long-term changes at the level of whole insect (and plant) communities, one using data from Great Britain and one from Germany. Oliver *et al.* (2015) analysed trends in frequency of occurrence (similarly to Powney *et al*, 2017, Section 1.1.1) of groups of British species from 1970 to 2009 according to their provision of key ecosystem functions. Among these, 720 species were grouped as providing pollination and included bees, hoverflies, moths, butterflies and as 'secondary' pollinators, wasps and two beetle groups. Overall, 27% of pollinating species were found to have declined (become less widespread) and 23% to have increased; these figures placed pollinating species together with groups providing pest control and cultural values as groups that had undergone significant net declines in GB during the period assessed (Oliver *et al.* 2015).

Hallmann *et al.* (2017) measured the total biomass of flying insects sampled using Malaise traps, deployed over 27 years in a series of nature reserves across Germany. They estimated a total overall decline in insect biomass of 76% between 1989 and 2016, although it is important to note that: 59% (37) of the locations were sampled in only one year during this period, and much of the biomass comprised of insects not generally important for pollination. Patterns at the species or insect group level are needed to help elucidate the reasons behind these dramatic changes in biomass.

### 1.1.4 Knowledge gaps

- Understanding of modelled species trends according to the ecological and functional traits of wild bee and hoverfly species, to help elucidate key drivers of both decline and increase or stabilisation in species occupancy, and opportunities for mitigation of declines.

- Patterns of bee and hoverfly species change at country, regional and habitat/ landscape level (a large number of repeatedly sampled sites as part of a national monitoring network would be required to detect changes at more local scales).

- A fundamental lack of data on changes in abundance of bees and hoverflies, stemming from a lack of long-term standardised monitoring (but see developments in Carvell *et al.* 2016). Linked with this, knowledge of the extent to which modelled species trends based on occupancy data reflect changes in local abundance and population size. Carvell *et al.* (2016) present a preliminary analysis using occupancy-detection models to combine occurrence data with systematically collected data for two wild bee species in a single year. This suggests that repeated systematic sampling from a network of sites not typically covered by recording schemes has the potential to improve both the precision of current occupancy estimates and fill gaps in the spatial extent of typical recording activity for some (especially widespread) species, as well as generating more standardised abundance estimates as are available for butterflies (and more recently, bumble bees).

## 1.2 Managed pollinators – Simon Potts

### 1.2.1 Honeybees

Vanbergen *et al.* (2014) reported a steady increase in the number of honey bee colonies from approximately 90,000 in 2008 to nearly 140,000 in 2012, with a parallel increase in beekeeper numbers from 15,000 to 27,000 over the same period. Since then, the overall UK estimated overwintering hive count figures (based on data extracted from BeeBase) have continued to increase (table 1.1); though the counts supplied by national beekeeping associations for individual countries (England, Scotland, Wales and Northern Ireland) are more variable across years (table 1.1)

Table 1.1: Estimated overwintering hive count data. * missing data. BeeBase (www.nationalbeeunit.com); BBKA, British Beekeepers Association (www.bbka.org.uk); WBKA, Welsh Beekeepers Association (www.wbka.com); SBA, Scottish Beekeepers Association (www.scottishbeekeepers.org.uk).

| Country | 2015 | 2016 | 2017 | Data source |
|---|---|---|---|---|
| UK | 198,711 | 223,087 | 247,461 | BeeBase |
| England | 117,493 | 132,173 | 140,668 | BBKA |
| Wales | 9,408 | 9,360 | 9,819 | WBKA |
| Scotland | 6,913 | 7,683 | 9,174 | SBA |
| Northern Ireland | * | 6,900 | 6,000 | |

However, there are high levels of uncertainty in estimates of honey bee hive numbers and the subsequent conclusions that can be reliably drawn because: (i) BeeBase is voluntary and may not reflect the full picture of beekeeping across the UK; (ii) for data to be reliable it is required that beekeepers ensure records are up-to-date. Initial investigation of BeeBase data shows there are several issues which require further investigation; and (iii) using estimated hives per keeper involves making assumptions based on a small number of responses and may not reflect the picture on the ground. Further work would be required to quality assure the approach. Despite these limitations it is currently the best source of data we have.

### 1.2.2 Managed bumble bees

Managed bumble bees, *Bombus terrestris*, are widely employed for commercial pollination in the UK. An estimated 40,000 – 50,000 colonies of the non-native subspecies (*B. t. dalmatinus* or *terrestris*) were imported annually to the UK in 2012 under importation licence WML-C22 (Natural England 2009, 2012 reported in Graystock *et al.* 2013). However, Class licence WML-CL22 was terminated at the end of 2014, and all growers and researchers planning to release non-native bumble bee colonies in glasshouses and polytunnels now have to apply for licence WML-CL28 and WML-CL34 respectively. The WML-CL28 grower licence can only be used in an emergency when commercial native bumble bees are not available and cannot be used for non-commercial situations, such as in gardens, allotments or orchards (Natural England 2018). The number of WML-CL28 licences issued has been low: 3 in 2016 and none in 2017 and 2018 (to date). Only 5 WML-CL34 research licences were issued from 2016 to date. The use of the native *B. terrestris audax* does not require a licence, and it is widely used in both glasshouses and open field situations, but no data on numbers is currently available.

### 1.2.3 Other managed species

The solitary trap-nesting bee *Osmia bicornis* is used for crop pollination on a very minor scale in the UK (compared to Europe), as is the common green bottle fly *Lucilia sericata*. No data on numbers are currently available.

### 1.2.4 Knowledge gaps

- Data on the total numbers of honey bee hives and beekeepers are based on voluntary records and so may not reflect actual numbers. Further, data is collected at the apiary postcode level and so does not provide an accurate representation of the spatial distribution of honey bee hives.

- Data on the current numbers of native *B. terrestris audax* imported are not available from licencing (not required) or from industrial suppliers. While this subspecies is native, its widespread use and release pose an unknown potential risk to wild native *B. terrestris* and other wild pollinators in the UK.

## 1.3 Wild plants - Claire Carvell

### 1.3.1 Wild plants

Vanbergen *et al.* (2014) presented an analysis of changes in range size and frequency of wild plant species associated with differing pollination requirements, using long-term data (since the 1930s) from the New Atlas of the British and Irish Flora (2002) and from the Countryside Survey of Britain. This revealed declines in insect-pollinated plants that were significantly greater than changes in non-insect pollinated wild plants over the same period, with declines being most pronounced in plant species dependent on more specialised species of pollinator (e.g. long-tongued bumble bees). The same Countryside Survey data were more recently used to assess historical changes in the availability of floral nectar resources for pollinators (Baude *et al.* 2016). This study estimated that the combined nectar resources from 260 common nectar-producing plants had declined in England and Wales between the 1930s and 1970s (an estimated 32% decline in national nectar provision per year). Total estimated nectar provision across Great Britain as a whole had stabilized by 1978 and increased from 1998 to 2007 (an estimated 25% increase), although not returning to the estimated 1930 levels for England and Wales.  The study also highlights a reduction in diversity of nectar sources up to and beyond 1990, and differences in the relative potential for nectar provision across different habitats, depending on their management and likelihood of flowering (for example, improved grasslands that are grazed but with a high proportion of *Trifolium repens*, white clover, have the potential to provide a greater nectar resource but this is contingent on changes in management approach) (Baude *et al.* 2016). This report is also discussed in section 4.1.2, containing more information on the relative contributions of different habitats to national nectar provision.

Since the 2002 New Atlas (Preston *et al.* 2002), the recording for which closed in 1999, no Indicator for changes in plant distribution has been published.  However, the Botanical Society of Britain & Ireland (BSBI) have continued to collect and collate much new data on plant distributions and approaches towards a plant indicator are being explored as described below. The average rate at which data have been added to the database over the period 2000-2018 is just over 1 million new plant records per year, resulting in a total of over 20 million records for this period. This brings the total number of plant records collated by the BSBI Distribution Database close to 40 million records. New change analyses are planned for the "Atlas 2020" project, for which a publication date of 2022 is expected. It is likely that any such analyses would use a variety of methods to produce trends, including Bayesian Occupancy Models (Isaac *et al.* 2014), as well as the previously used Telfer Change Index (Preston *et al.* 2002; Vanbergen *et al.* 2014). Other options for a national plant indicator have been explored by CEH in recent years under indicator development contracts from JNCC.

The National Plant Monitoring Scheme, launched in 2015, is another source of plant data. This scheme currently surveys between 400 and 450 one kilometre squares per year, across the UK, Isle of Man, and the Channel Islands. A technical review of this scheme is currently under review for publication as a JNCC technical report (Pescott *et al.* In review). Indicator development work that has been carried out by the NPMS partnership includes the construction of Bayesian spatio-temporal models for species richness, as well as habitat-specific trend lines. In general, data resources and reports from the scheme can be found at http://www.npms.org.uk/content/conservation-and-research. Furthermore, the network of 1km survey squares that forms the UK Pollinator Monitoring Scheme is co-located with randomly selected NPMS survey squares across England and Scotland, allowing for future analyses of parallel changes in pollinators and plants at the same sites.

### 1.3.2 Knowledge gaps

- Understanding of the role of different insect species (including, for example, the Lepidoptera) as pollinators of wild plants, and whether changes in insect community composition and overall abundance are affecting the reproductive success and population sizes and/or distributions of insect-pollinated plants.

## 1.4 Pollination services and pollinator dependent crops – Mike Garratt and Tom Breeze

### 1.4.1 Pollination services

The status and trends in pollination services depends on both how the demand for pollination service by crops and wild plants is changing in space and time as well as any change in the capacity of flower visiting insects to deliver these pollination services. None the less, the value of more diverse pollinator communities in providing improved pollination service is clear (Hoehn *et al.* 2008, Kremen and Winfree 2009, Winfree *et al.* 2018). The reported declines in occupancy of many pollinator species (Figs 1.1, 1.2) could indicate risks to pollination of crops and wild plants. Pollination service delivery however, is driven by visitation rate (Garratt *et al.* 2014a, Garratt *et al.* 2016), which is a function of pollinator abundance. Most crops rely on common widespread species for pollination (Kleijn *et al.* 2015).  Without good data on changes in pollinator abundance in space and time, any inferences about changes in pollination services have limitations (Carvell *et al.* 2016).

Significant advances in our understanding of which pollinators are providing pollination services to crops across Europe and North America have been made (Kleijn *et al.* 2015), and while some pollinator species are particularly common crop flower visitors overall, different pollinator communities are important for the pollination of different crops (Garibaldi *et al.* 2015, Garratt *et al.* 2014a). Such findings allow for better quantification of changing pollination service in the face of possible pollinator species, or functional group decline. Furthermore we have begun to characterise to what extent crop dependence on insect pollination varies between crop varieties (Garratt *et al.* 2016, Hudewenz *et al.* 2014) allowing for more nuanced estimates of the changing status and trends in crop pollination demand in the UK. Although data for many crops and varieties is still missing. In addition there is increasing evidence of the influence of context on crop pollinator dependence for many crops including reported effects of soil fertility (Garratt *et al.* 2018, Klein *et al.* 2015, van Gils *et al.* 2016), climate (Bishop *et al.* 2016) and other ecosystem services (Bartomeus *et al.* 2015, Lundin *et al.* 2013) making us better able to predict changes in demand for pollination service.

Pollination deficits of around 20% have been identified in some apple varieties (Garratt *et al.* 2014b) and probable pollination limitation has been found in beans and oilseed rape (Garratt *et al.* 2014a) but there is still a clear lack of research investigating the extent of pollination deficits across crops and regions in the UK. A few recent studies have specifically investigated how crop pollinators or crop pollination services may change over time in the UK. Woodcock *et al.* 2016 found increased local extinction rates for species found foraging on oilseed rape. This was attributed to exposure to neonicotinoid insecticides and has implications for the pollination of oilseed rape in the face of these declines (see section 3.2.1 for further details of this study). In another study the impact of future climate scenarios on the distribution of crop pollinators and the optimal environmental conditions for growing a key UK crop, apples, were compared under a projected future climate in 2050.  A clear mismatch between pollinator distributions and optimal crop area was found indicating that pollination service to certain crops may be more vulnerable in the future (Polce *et al.* 2014). Such studies investigating changing status and trends in pollination service are still few and far between.

## 1.4.2 Pollinator dependent crops and pollination capacity of honey bees

Between 2010 and 2015, the area of insect pollinated crops in the UK has increased by approximately 2%, and has been generally consistent, with some fluctuations depending on trends in insect pollinated arable crops (oilseed rape and field beans). There is no universal measure of demand for pollination services however, using literature on the recommended densities of honey bee colonies required to pollinate one hectare of each insect pollinated crop as a proxy (Breeze *et al.* 2014), the UK demand for pollination services is approximately 1.8M hives over the year. Following the methodology of Breeze *et al.* (2011) and with new estimates of hive numbers supplied by the National Bee Unit, the capacity of the UK's honey bee stocks to supply pollination services is, as of 2015, only sufficient to meet approximately 30% of this demand, even if all UK hives were to be used for pollination in optimal conditions (figure 1.3). These estimates are highly tenuous however as not all hives are registered with the National Bee Unit and this data is not available for every year in consideration (following Breeze *et al.* 2011, it is assumed that stocks remain constant if data is unavailable). Also, due to a lack of pollination service monitoring, the actual extent of honey bee pollination services remains unknown, but is likely to be significantly lower due to the small market and low payments for honey bee pollination services (Breeze *et al.* 2017).
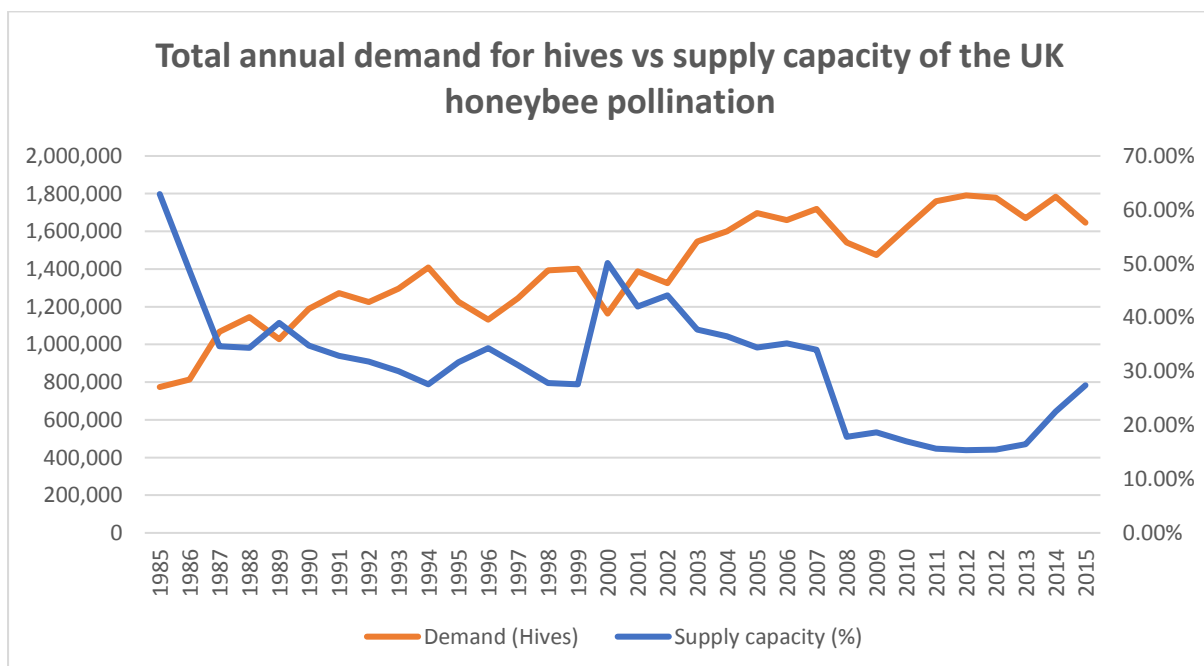


**Figure 1.3:** Demand = the annual number of honey bees hives required, throughout the year, to provide recommended levels of pollination services to all insect pollinated crops (excluding those grown in glasshouses). Supply capacity = the capacity of the UK's honey bee stocks, under optimal pollination management, to provide pollination services to UK crops. Note that years with no data on honey bee numbers are assumed to have the same number of hives as the previous year.

## 1.4.3 Knowledge gaps

* **The scale of managed pollinator use within the UK.** Despite its importance as an input in crop production, the management of pollination by agribusiness has not been measured to the same extent as pesticide, fertilizer or other inputs. Combined with limited information on the number and distribution of honey bee hives, this lack of data makes it impossible to accurately estimate the extent of managed pollinator use throughout the UK. As pollination contracts for professional

beekeepers are uncommon and remain relatively low in price relative to the expenses of beekeepers (Breeze *et al.* 2017), a better understanding of how farmers use wild and managed pollinators would help facilitate links between farmers and beekeepers and provide an insight into how resilient UK farming would be to losses of managed pollinators (e.g. import restrictions on bumble bees or severe honey bee losses).

- **Variation in dependence on pollination between crops and varieties.** Spatial estimates of crop pollination demand are typically based on a single dependency measure for each crop.  However emerging evidence highlights how this level of dependence varies between crop varieties (e.g. oilseed rape has varieties with 5-50% dependence).  We have some data on different varieties of apple and oilseed rape but for many crops and varieties lack research.

- **Changing abundance in key crop pollinators.** Sufficient pollination to both crops and wild plants requires the minimum number of sufficiently effective flower visits to be met.  This is not determined simply by the presence or absence of suitable pollinators (which is the output of most models to date) but is also dependent on adequate <u>abundance</u> of pollinators.  Without good data on changing abundance of pollinators it remains a challenge to make any estimates of the changing status or trends in pollination services.

- **Existing levels of pollination services.** Although yields of many insect pollinated crops have risen (Aizen *et al.* 2008), this does not in itself indicate stable levels of pollination services. Other factors, such as new varieties and improved crop planting systems (e.g. enclosed polytunnels for cherries) could also create sufficient yield increases to mask deficits from pollination services. Nonetheless, production deficits of approximately £6000/ha due to sub optimal pollination have been observed in Gala apple orchards (Garratt *et al.* 2014b). Without systematic monitoring and reporting, it is not possible to identify areas of the UK which are likely to be experiencing pollination service deficit, or estimate the economic impacts this has. Understanding the scale of these deficits would facilitate better conservation targeting and stronger assessment of the costs and benefits of management options for key pollinators.

- **Past, present and future levels of wild plant pollination.** While there are limits in our understanding of current and changing levels of crop pollination and deficits, our knowledge of wild plant pollination is even sparser. We have information on the dependency of some plant species, but little or no understanding of the extent of deficits. We know that pollen/pollination limitation is common in plant species (evidence from global meta-analyses by Ashman *et al.* 2004) so we might expect the same for the UK, but this has not be measured.

## 1.5 Monitoring status and trends of pollinators using DNA technologies - Hannah Norman

### 1.5.1 Introduction

UK scale monitoring of pollinators requires the collection, processing and analysis of extensive numbers of samples. There is potential for new and developing molecular methods to improve the cost and time efficiency of processing and analysing specimens, which may enable more efficient monitoring. Brief descriptions of molecular methods, their requirements and uses, and associated gaps in knowledge are given in the summary below. In order to monitor status and trends for pollinators across the UK, sample collection from a large enough network of sites is required. Carvell *et al.* (2016) found that the number of sites needed depends on the abundance of pollinating species and the level of change the monitoring needs to detect. To detect a 30% change over ten years in groups with ten or more individuals at each site, 20 to 75 sites are needed. However, for smaller groups or to detect a smaller amount of change, 145 sites selected across the UK would be needed

to detect changes in species diversity and abundance. In 2015, 14 sites were surveyed with pan traps and transects, giving a large amount of data to process (Carvell *et al.* 2016). Currently the identification of pollinators is carried out using morphological taxonomy. In the pilot study this was done by professionals, and in many studies insects are collected and returned to the lab for identification (Nayak *et al.* 2015). Identifying pollinators in this way is time consuming, and is challenging for non-experts. Due to the diversity of pollinators and the difficulty of separating many of the species, non-experts are limited to identifying pollinators to higher taxonomic levels. Using DNA techniques for monitoring populations is an emerging field, with different methods suitable for a range of different sample types. These methods differ in how they analyse the samples, what information they produce and how much they cost (Tang *et al.* 2017). Here, several different methods are explained in the context of pollinator monitoring and their potential suitability. DNA methods have not been widely applied to pollinator monitoring, especially in the UK, hence there are large knowledge gaps. DNA methods are considered for monitoring programmes with relatively low confidence due to the lack of evidence in the literature.

## 1.5.2 Molecular methods

In the past, molecular methods have been costly, with the cost increasing exponentially the more specimens processed. The early Sanger sequencing method sequences each specimen individually, meaning that the cost of each sequence is static at around £9, no matter how many samples there are (Tang *et al.* 2017). This has made DNA methods unsuitable for the large datasets obtained from pan traps in monitoring programs. More recently, next generation sequencing methods, most commonly Illumina sequencing, have changed the way in which DNA can be utilised. This method allows high throughput sequencing of thousands of samples at once, producing large volumes of data suitable for sequencing samples containing multiple species. The initial cost for sequencing using this method is high, but with each sample added the cost per specimen decreases, meaning that for datasets containing more than a few hundred individuals, this method is more cost effective (Tang *et al.* 2017; Stein *et al.* 2014; Carvell *et al.* 2016).

***Metabarcoding:*** There are different ways of processing samples using next generation sequencing. The most common method is metabarcoding, where mixed samples are kept together and a barcoding gene is amplified for all specimens in the sample. This enables the species composition of samples to be established at a low cost (Stein *et al.* 2014). However, there are biological and technical biases in the method which limit the ability to provide abundance data (Pawluczyk *et al.* 2015; Elbrecht and Leese, 2015). Currently this makes it an unsuitable method for pollinator monitoring, although quantifying these biases for different species and applying corrections during analysis may provide reliable abundance estimates for mixed samples (Thomas *et al.* 2016). In future this could lead to metabarcoding being suitable for pollinator monitoring.  Abundance measures are needed to detect a proportion of change in a community. If only species diversity is used, then change would only be detected at the point where species were being lost in a community, and thus would not detect the lower levels of change that give an earlier indication that a community is changing.

***Illumina Barcoding:*** A method termed Illumina barcoding was trialled on bee and hoverfly specimens from across the UK by Tang *et al.* (2017), and was able to successfully identify almost all UK bee species. Here each of the bee and hoverfly specimens was sorted and tagged, allowing the sequence to be traced back to the specimen it came from. Importantly, this method of DNA monitoring allows abundance to be measured because all of the specimens are kept separate.

## 1.5.3 Reference databases

Reference databases are vital to using DNA for monitoring, because they provide the link between a sequence and a species (Hebert *et al.* 2003; Taberlet *et al.* 2012). This is similar to a key for morphological identification, as it provides a known sequence for each species of interest, against which unknown sequences can be matched and identified (Hebert *et al.* 2016; de Vere *et al.* 2012). Often these reference libraries are publically available online, and have been used in studies to identify species such as the economically important pest, *Brassicogethes aeneus* (Ouvrard *et al.* 2016), from pan traps. For insects the CO1 (cytochrome oxidase I) barcode is most widely used for species identification. This is a mitochondrial gene which has been shown to have enough power to distinguish between bee species (McKendrick *et al.* 2017; Gonzalez-Vaquero *et al.* 2016). A COI database for UK bees has been developed and is now over 90% complete (Tang *et al.* 2017). This database is able to distinguish almost all UK bee species, showing that COI is an appropriate barcoding gene for this group. COI barcode databases are also being developed for bees in countries around the world, including Germany (Schmidt *et al.* 2015), Chile (Packer *et al.* 2016) and Canada, where there is currently a database of around 95% of the countries bee species (Sheffield *et al.* 2017). Whilst reference databases for bee species are being established in many countries (Sheffield *et al.* 2017; Schmidt *et al.* 2015; Packer *et al.* 2016), there is a lack of projects developing reference databases for other pollinating groups. Projects to develop barcodes for UK species of hoverflies, butterflies and moths would all increase the potential of DNA monitoring in the UK. This adds an initial cost to DNA monitoring, and once the resource is developed it should be maintained to keep the database up to date with new species arriving in the UK. This could be a challenge as research staff often move on from projects and institutions, however there is the potential to utilise natural history collections for this, as they already host databases of species. Museums house collections of pollinating insects which could provide a resource for developing the reference databases, matching barcodes to well-curated museum specimens for a reliable reference database (Miller *et al.* 2016). As well as this, data mining of online sequences can provide more references from different research projects and different countries with the same species (Huemer *et al.* 2014). These need to be curated to ensure they are reliable but can decrease the time and effort needed to develop the reference database.

## 1.5.4 Comparing morphological techniques with DNA monitoring

Tang *et al.* (2017) compared identification by taxonomist verses DNA by identifying the same specimens using both methods. It appeared that taxonomist identifications were subjective, with different individuals possessing different error rates and making different mistakes. Taxonomists misidentified 7.2% of the specimens, whereas DNA identified none incorrectly. However, DNA identified fewer specimens to species level than taxonomists (8.1% verses 4.3%). When considering just bees, which had an almost complete reference database, this dropped to 2.2%, suggesting that in this case the main barrier to molecular identification is the reference database. Molecular methods are not well established in monitoring. The most well-known use of DNA in UK monitoring is the use of environmental DNA (eDNA) for detecting the great crested newt (Biggs *et al.* 2015), which is an efficient and reliable way of detecting presence. A more relevant use of DNA for monitoring is in the metabarcoding of freshwater invertebrates, which, like pollinators, are a large diverse group which are challenging to identify using morphology. Bista *et al.* (2017) found that metabarcoding could be used to monitor the diverse group of Diptera found as larvae in freshwater lakes, giving species diversity estimates over time which matched with traditional diversity measures. There is however a lack of well-established molecular monitoring programs to compare to the potential DNA monitoring of pollinators.

### 1.5.5 Monitoring of networks

Pollinators do not live in isolation, but contribute to ecosystem functioning through pollination of crops and wild flora, as well as interacting with other elements of the ecosystem such as parasites and disease, which may impact their functioning.  Monitoring of species diversity does not give an insight into these complex networks, and gives a simplified picture of pollinator functioning and health. The techniques outlined above result in a large amount of genetic data, and so give the capacity to detect associations with individuals as well as a species identification. This opens up the opportunity to monitor floral visitation from pollen sacs (Wilkinson *et al.* 2017; Galliot *et al.* 2017) and honey (de Vere *et al.* 2017; Hawkins *et al.* 2015), as well as other associations such as parasites (Jabal-Uriel *et al.* 2017). These areas may not be suitable for large scale monitoring but may be able to provide a more in depth view of pollinator population health and network interactions. This gives a more complex picture of how different species and individuals are interacting within the environment, which could help focus management efforts to maintain a diverse group of pollinators which are effective pollinators of the plant community.

### 1.5.6 Knowledge gaps

- There are other methods that can be used to sequence bulk samples of specimens, such as mitochondrial metagenomics (Tang *et al.* 2015; Crampton-Platt *et al.* 2016; Andujar *et al.* 2015; Gomez-Rodriguez *et al.* 2015), and these have the potential to give abundance data. This is currently not realised due to issues with differences in body size which mean that it is difficult to work out relative and absolute abundance. Resolving this this could lead to a DNA monitoring method which removes the need to sort samples and still gives abundance. This would reduce the time needed to process the samples as there is no sorting step, and no time consuming PCR is required. Currently post sequencing bioinformatics processes are intensive for this method and can take a long time, and so future research would have to also simplify and streamline this stage of the process.

- Whilst there is some information on the cost of different DNA methods (Carvell *et al.* 2016), there has not been a recent cost analysis of these different methods and how they compare to the cost of traditional monitoring with morphological identification for pollinator monitoring in the UK. This would allow a better assessment of the feasibility of incorporating DNA methods into a national monitoring framework

- As well as cost analysis there is a knowledge gap in how DNA methods could be integrated into a national monitoring framework. There may be logistical issues with collecting and storing samples for DNA analysis, especially as the majority of UK monitoring relies on volunteers. Getting volunteers to collect samples for this method may have different challenges which have not been considered.

- In order to use DNA methods for monitoring of all UK pollinators, a curated reference library of barcodes for other pollinators as well as bees is needed. There is the potential to focus on indicator species at first, to develop a useful resource for pollinator monitoring which could be used in a monitoring framework.

- Monitoring pollinators using DNA is an area which is beginning to be explored. These methods and resources are much more developed for bees, both in the UK (Tang *et al.* 2017) and abroad (Schmidt *et al.* 2015; Packer *et al.* 2016; Sheffield *et al.* 2017), than for other pollinating groups. The large diverse groups of pollinating insects, which are often difficult to identify, potentially make good candidates for DNA monitoring. However, there are several different methods which give differing amounts of data, making it important to choose the method carefully. Abundance can be measured using Illumina barcoding, however this increases the time and cost of the process. The more straightforward metabarcoding method is cheaper but does not give abundance measures. It is important that cost analysis is carried out on these methods in the context of UK pollinator monitoring, before they can be shown to be a practical addition to current monitoring of UK pollinators.

# 2. Values of pollinators
Author: Tom D Breeze

## 2.1 Economic Impacts of pollinator populations – Tom Breeze

### 2.1.1 Overview

By influencing crop production, pollinator populations can have considerable economic impacts for farmers and consumers around the world. These impacts can range from direct economic <u>benefits</u> to productivity to more intangible economic <u>values</u> to consumers (by maintaining lower prices for pollinated foods) and producers (through increased profits) alike (see IPBES, 2016 for a full discussion). A number of methodologies have been used to estimate these impacts (see IPBES, 2016 and Hanley *et al.* 2015 for a summary), but have yielded very inconsistent results (Breeze *et al.* 2016). Although estimating these economic impacts is thought to be useful for informing wider landscape management, very little of the global literature is actually useful for practical decision making and only provides a limited perspective on the full impacts of pollinators on the economy and society (Breeze *et al.* 2016). This section presents an update on the economic impacts of pollination services from 2014 and a concise overview of the knowledge gaps that currently limit research capacity to provide more applied advice to stakeholders.

The previous report published in 2014 by Vanbergen *et al.* presents an overview of the theory behind economic valuation of pollination as an ecosystem service and provides a first, very basic estimate of economic benefits to production of £603M for the year 2010. Since this publication, a number of studies have built upon the methods described to provide new insights into the economic benefits of pollination services to the UK. These studies are discussed and critiqued below.

### 2.1.2 Economic impacts of pollination services in the UK

Pollinators can have economic impacts to different people across the food chain as well as across time and space (summarised in table 2.1). To date, estimates of impacts in the UK have focused largely on the economic benefits of pollination to production, the total market price of the additional output resulting from pollination services, or to farmer profits for a single, specific crop (Garratt *et al.* 2014; 2016).

**Table 2.1**: Economic benefits of pollination services (with example references concerning the UK)

| Benefits | Description | Examples |
|---|---|---|
| Market production | Pollination directly increases the quantity and quality of yield in many crops. This benefit can be quantified by estimating the differences in total crop output (including quality) with and without pollination at current market prices. | Smith *et al.* (2011), Ollerton *et al.* (2016)* |
| Producer profits | Many of the costs for farmers are static regardless of the yields they eventually produce. By increasing production, farmers can gain a greater degree of net profit, however some costs can also increase such as the labour costs for picking/thinning fruits. | Garratt *et al.* (2016; 2014)* |
| Waste reduction | Pollination can increase the storage life of some crops, reducing the economic costs for farmers and supermarkets in managing waste. | Wietzke *et al.* (2018) |
| Consumer surplus | By increasing the supply of a crop relative to demand, pollinators help reduce the costs for consumers. This gap between the price that is paid and that price that would be paid if supplies were at levels without pollination services can be estimated with economic models. | Bauer and Wing (2016) |

| Costs avoided | Alternatives to wild pollination services are available, ranging from additional paid pollination services to mechanical methods of pollination. By providing this service for little or no cost, wild pollinators effectively save farmers this expense. | Allsopp *et al.* (2008) |
|---|---|---|
| Resilience | Strong, diverse pollinator populations can produce more reliable and stable on farm production across years and buffer farmers against unexpected impacts from weather events. | None to date. |
| Natural Capital | By increasing crop productivity, pollinators, like many natural resources, are a potential input into agricultural systems. This input is not always utilized (e.g. if an arable farmer is growing a non-insect pollinated crop in between insect pollinated crops) but still has potential economic value which can be measured independently of any benefits actually being received. | Ricketts and Lonsdorf (2013) |

*= UK example

Almost all research into the economic benefits of pollination services has focused on estimating the economic benefits to market production and producer profit (Breeze *et al.* 2016). This is usually done by using scientific literature to estimate the proportion of crop production resulting from pollination and calculating the market price of this production using these "dependence ratios" (see IPBES, 2016 and Hanley *et al.* 2015 for more details). An updated version of these figures from Vanbergen *et al.* (2014) is presented in table 2.2 (figure 2.2 reflects these changes over time), however numerous studies have highlighted the weakness with these estimates (Hanley *et al.* 2015; Garratt *et al.* 2014a; Breeze *et al.* 2016), which are discussed below.

Table 2.2: Crop dependencies on pollinators and estimated UK market economic benefits of pollination in 2015 (most recent year available)

| Crop | Dependence on Pollinators | Market sale price (in £M) 2015 | Pollination benefits (in £M) 2015 |
|---|---|---|---|
| Oilseed Rape | 25% | 720 | 176 |
| Strawberry | 45% | 284 | 128 |
| Dessert Apples | 85% | 77* | 65* |
| Raspberries | 45% | 123 | 56 |
| Culinary Apples | 85% | 40* | 34* |
| Tomatoes | 25% | 119 | 30 |
| Runner Beans | 85% | 15 | 13 |
| Pears | 65% | 13 | 8 |
| Plums | 65% | 11 | 7 |
| Other | 5-85% | 229 | 98 |
| Total | | | Approx. 615 |

Note: updated from the previous version in Vanbergen *et al.* (2014). Cucumbers have been removed from this analysis as the UK industry relies on self-pollinating varieties due to demands for seedless vegetables. * = Due to the price data used in these calculations, these figures are significant underestimates (see Garratt *et al.* 2014).
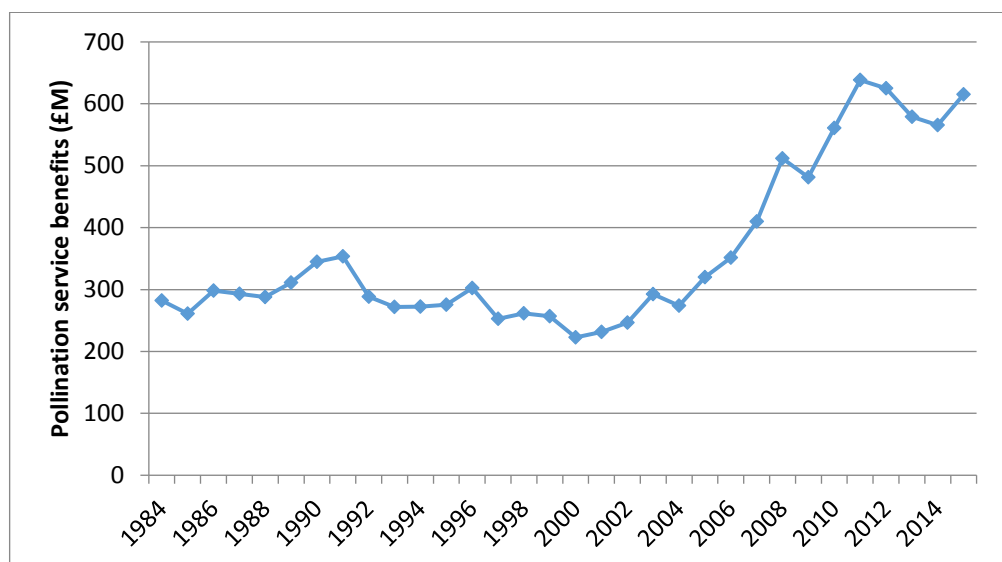
**Figure 2.1:** Annual estimations of the economic benefits of pollination services to crop production in the UK (in £M) 1985-2015

These figures are unlikely to be accurate for several reasons. First, they do not account for the impacts of pollination services on producer costs and, by extension, net profit. In apples, data collected by Garratt *et al.* (2014) indicate that without pollination, labour costs per hectare would fall by approximately 60% due to lower thinning requirements, however this saving is trivial compared to the loss of productivity without pollination. Nonetheless, such changes have an effect on the net profit benefits from pollination and remain unknown for many crops.

Secondly, we do not know the level of pollinator dependence of many varieties of crops grown within the UK. The dependence scores in figure 2.2 are generalizations of global literature, much of which is very old and does not pertain to the UK.  In reality, several studies (e.g. Garratt *et al.* 2016; Hudewenz *et al.* 2014) have observed that the benefits of pollination services varies across varieties of the same crop. For example Garratt *et al.* (2016) observed that, by increasing both yield and quality, pollination services increased net profit of 4 apple varieties by between 48% and 68%.

Third, because available yield data does not include the proportion of crops from different quality grades the figures in table 2.2 only account for a change in yield, not quality. As quality increases from pollination can play an important role in the sale price of a crop, capturing these impacts is essential to properly measure benefits (Garratt *et al.* 2016; Klatt *et al.* 2014). To illustrate this, Garratt *et al.* (2014) explored the impacts of failing to account for crop quality and variety specific effects of pollination on estimates of economic benefit in two of the most widespread apple varieties. Their analysis indicates that these discrepancies produce an £8.8M underestimate of the benefits of pollination services. This may be the case in other high-value crops that are quality driven.

Finally, these figures assume that pollination services are at optimal levels already. In reality, Garratt *et al.* (2014) have already observed pollination service deficits in gala apples, where approximately £6M of production is already being lost due to inadequate pollinators. Similarly, in some crops, over-pollination may result in additional costs, such as increasing the costs of thinning apples or the reduction of apple quality (Garratt *et al.* 2014). Without monitoring of pollination services, it is not possible to estimate how widespread this effect is but it is thought to be likely due to the pressures put on many pollinator populations.

## 2.1.3 Shortcomings with Economic valuation

Although economic valuation has the potential to inform management at a local or even national scale (see Knowledge Gaps 3, 4 and 5), a strictly economics approach risks missing many other valuable outputs from pollinator populations. Firstly, as many of the UK's rarest pollinators are not crop pollinators and require specific management, emphasising management for economically significant crop pollinators risks neglecting these species (Senapathi *et al.* 2015; Kleijn *et al*, 2015).  Furthermore, humans naturally associate value with the aesthetics and utility of the landscape itself (Junge *et al.* 2015). Several studies have demonstrated that a large component of this landscape appreciation is the presence and diversity of flowering plants and rare habitats within the landscape (Junge *et al.* 2009; 2015). Due to our limited understanding of pollination networks across the UK, focusing conservation efforts exclusively upon pollinators which are significant for crop production risks weakening pollination services to other plants within the landscape. This in turn may threaten floral biodiversity as a majority of plants depend upon pollinators to reproduce (Ollerton *et al.* 2011). By considering these biodiversity and social values as well, solutions which are economically sub-optimal but which maximise other benefits, may be identified and pursued over economically optimal options (Senapathi *et al.* 2015).

## 2.1.4 Knowledge gaps

Ideally, economic measurements of pollination services should facilitate better management of biodiversity and ecosystem services by i) highlight long and short-term human impacts of the service to a range of stakeholders (table 2.1), ii) identifying land which provides beneficial pollination services that would otherwise go overlooked, iii) identify the benefits of management measures to promote pollinators. To date, almost all published literature, including this document, has only provided a basic estimation of the market benefits of pollination services to crops. While such figures are useful for illustrating benefits to broader audiences, they are not useful for informing long-term management (Breeze *et al.* 2016). Although interdisciplinary methods are available to provide more applied estimates of benefits, their use is limited by a number of key knowledge gaps:

1. **Level of dependence of contemporary varieties of different crops on pollination services.** As stated above, there is little information on how different varieties of crops benefit from insect pollination, with the information that is available stemming from research projects (Garratt *et al.* 2016; Hudewenz *et al.* 2014). As farmers will often change what varieties they grow, depending on market conditions, disease resistance etc., this knowledge can become obsolete very quickly. This makes it difficult to advise on how important pollination actually is to crops. Assessments of pollination service benefits in different crop varieties can be done relatively simply using existing methods (Garratt *et al.* 2016; Carvell *et al.* 2016), ideally as part of standard assessments in e.g. oilseed rape varieties on the HGCA recommended list. This would provide growers with an immediate measure of the importance of pollination services and provide researchers with information on pollinator dependence under ideal conditions without the need for dedicated research projects.

2. **How the economic benefits of pollination vary between different pollinator communities.** Although recent studies have identified the key pollinators of some major crops in some parts of the country (Garratt *et al.* 2014b; 2016), we do not yet know a) how these communities can vary across the country and b) the exact relationship between the number of visits by different species and economic output. Methods for assessing typical crop visitor communities are well established (Garratt *et al.* 2016; Carvell *et al.* 2016) but would need further effort to apply across the country and to the full range of insect pollinated crops. Understanding these factors would allow us to identify what species farmers should focus their management efforts on to maximise the benefits of pollination services.

3.  **Understanding the distribution and value of pollinator natural capital across the UK and identifying areas of possible deficit.** Pollinator populations are a form of natural capital, an economically important natural resource that has the potential to be used by farmers to increase output. Although ecological models can be used to predict these populations (Haussler *et al.* 2017; Koh *et al.* 2016), these models have only been validated against a small amount of field data, so their accuracy is still uncertain. Understanding the distribution and potential economic value of this natural capital would facilitate better spatial planning of resources and help identify high-value areas within a landscape (e.g. Ricketts and Lonsdorf, 2013). Combining this with better information on pollinator dependence (Knowledge gap 1) and pollinator community composition (Knowledge gap 2) would also allow projections of the long-term resilience of pollination services to changing climate, market or land use conditions. This knowledge gap is currently being addressed by the Modelling Landscapes for Resilient Pollination Services project funded under the Global Food Security Programme and expects to deliver results in 2021.

4.  **The economic costs and benefits of different management options for pollinators and how they can be made more cost-effective.** Although a considerable body of research has demonstrated that tailored, well managed agri-environment measures can bolster pollinator populations and lead to improved yields (e.g. Pywell *et al.* 2015), very little work has directly assessed the links between increased pollinator populations and increases in crop output. Field methods (e.g. Garratt *et al.* 2016) and yield models (Garratt *et al.* 2018) suitable to undertake this have been developed but are not yet widely applied. If maintained over a long enough time period (5 years or more), this would facilitate farm level cost:benefit analyses of management options to farmers (e.g. Blaauw and Isaacs, 2014), which may provide incentive to undertake many currently unpopular management options (Breeze *et al.* 2014b). If combined with maps of pollinator natural capital (Knowledge gap 3), addressing this knowledge gap would also facilitate targeting of agri-environment funds to ensure cost-effective spending in the long-term.

5.  **Level of resilience of UK food systems to pollinator changes, nationally and abroad.** Although the yield benefits of pollination services to crop yield and quality have been established for a number of UK crops (Garratt *et al.* 2014 a, b, 2016) comparatively little is known about how losses of pollinators would affect other stakeholders within the supply chain. For example, reduced supplies of popular crops could drive up prices for consumers and processors or force them to seek alternatives (Bauer and Wing, 2016). There is also emerging evidence that pollination services may reduce waste by increasing the shelf-life of some crops post-harvesting, which would have further impacts on the costs of these crops to consumers, but this would require further lab and economic analysis to fully quantify the economic benefits. Addressing this knowledge gap would require more detailed data on where crops produced and imported into the UK are bought and sold to and how shifts in supply ripple through the supply chain (e.g. Bauer and Wing, 2016).

## 2.2 Social and cultural values of pollinators and pollination – Tom Breeze

### 2.2.1 Overview

In addition to their overt economic market impacts (table 2.1), pollinators and animal pollinated plants contribute to a wide range of socio-cultural values (table 2.3). The IPBES assessment on pollinators (IPBES, 2016) recognises and reviews these values from across the world, highlighting the direct and indirect importance of pollinators and pollination to health, education, heritage and wellbeing. However, very little of this literature comes from urban people or the developed world, instead focusing on traditional and indigenous people in developing nations where local communities and heritage has a stronger connection to nature.

Table 2.3: Overview of socio-cultural values associated with pollinators and pollination services

| Benefits | Description | Examples |
|---|---|---|
| Option value | Option values cover the economic and non-economic benefits that have not yet been discovered. By supporting plant diversity, pollinators maintain these options for future use e.g. new plant-based medicinal compounds. | None (but see Ollerton et al. 2011, for plant diversity impacts) |
| Aesthetic values | Insect pollinated plants include many vibrant flowering plants that affect the visual quality of the landscape for different people. | Akbar et al. (2003), Junge et al. (2015) (Swiss example) |
| Cultural symbolism | Bees and other pollinators as well as the plants they pollinate can have symbolic value, acting as symbols of nations or ideals | Ollerton et al. (2016). IPBES (2016 – International review) |
| Existence value | Existence value represents the innate value that people place on something simply because it exists and they do not wish to see it become extinct. This is true for bees and a number of other animals and pants | Breeze et al. (2015), Mwebaze et al. (2018) |
| Health values | Insect pollinated crops are key components of human diets around the world. In the absence of pollination, lower availability of these nutrients could result in greater incidence of severe preventable diseases | Smith et al. (2015) |

To date, only one published study that we are aware of, has quantitatively explored these values in the UK; Ollerton et al. (2016) indicated that public willingness to pay for holly and mistletoe decorations with berries (which are entirely dependent upon insect pollination) were more highly prized at auction, selling for double to triple the price of berry-less plants. The plants and their berries also have cultural significance in Christian and pagan mythology (Ollerton et al. 2016). Additionally a few studies have used economic methods to try and quantify the societal value of bees (Mwebaze et al. 2018) and the aesthetic value of pollinated wildflowers (Breeze et al. 2015). All of these studies highlight a high willingness of the public to support the preservation of bees, even aside from their other societal benefits (Breeze et al. 2015). However, these studies are based on stated preference surveys, which present respondents with a hypothetical monetary market where they are asked to pay for pollinator conservation. As such, they only capture a very particular expression of the "value" of pollinators and pollination which is tied to the monetary instrument and the scenarios in question. This approach suggests that the British public believe that tax money (£349M-£1.1BN) should go towards the preservation of pollinators but little more.

Exploring these social and cultural values using more suitable metrics could however provide a more complete picture of the trade-offs in decision making and environmental management (Senapathi et al. 2015), as well as fostering further understanding about the role of pollinators, and ecology in general, in the UK.

Research into the social and cultural values of pollinators has been carried out in England, using interviews, discourse analysis, art ethnography, and workshops (Christmas et al, 2018). Collectively, the results indicate

that people value pollinating insects in many ways: as beautiful or fascinating creatures; as providers of goods; as managed animals and as animals with character of their own. However, participants most often adopt the idea of pollinating insects as being part of a more interconnected system of living beings, including humans, rather than simply as providers of a service (honey or pollination). They tended to approach the idea of pollinating insects as providers of the pollination service for humans without commitment, and quote that they ought to value pollinators as inherently useful things. This can inform public engagement around pollinating insects, and is also a useful basis for assessing the public's changing attitudes towards, and awareness of, pollinators (Christmas *et al*, 2018).

## 2.2.2 Knowledge gaps

1.  **The pollinators and pollinator dependent plants in the UK which have cultural value.** Although the symbolic value of certain plants has been documented for centuries, there is no comprehensive database on plants or pollinators that are culturally significant in the present day. A full review of the cultural importance of pollinators and animal pollinated plants would give a valuable insight into the wider values that the public hold and how this can be used to raise awareness and measure impacts from real-world changes. Ideally this would cover multiple forms of cultural values, including recreation, aesthetics (e.g. Junge *et al.* 2015 from Switzerland) and symbolism (e.g. Ollerton *et al.* 2016 from the UK) and should capture contemporary culture (e.g. cinema) which can shape our understanding of these values.

2.  **How much of the UK's flora is dependent upon insect pollination.** Although we know that the majority of flowering plants are insect pollinated (Ollerton *et al.* 2011), there is no specific estimate for how much UK flora depends on insect pollination. Integrating this knowledge with existing floral biodiversity records and regular surveys of the extent of habitats (e.g. the Countryside Survey), would allow mapping and monitoring of the trends in socially and culturally significant plants and habitats, as well as identifying future risks.

3.  **Level of importance of pollinated crops for health and wellbeing in the UK.** In addition to the economic benefits of crop production, a number of recent studies have highlighted the role of insect pollinated crops in maintaining healthy diets and avoiding preventable diseases (Chaplin-Kramer *et al.* 2014; Smith *et al.* 2015). Currently the UK health service faces pressure from a large number of diet related illnesses, particularly obesity. Many of these issue are linked with low intakes of fruit and vegetables which may be exacerbated by lower access to/higher prices (Tiffin *et al.* 2011; Jones *et al.* 2014) in the absence of pollination. Understanding the health benefits from a strong supply of pollinated crops would provide further insight into the benefits of widespread conservation initiatives.

4.  **The extent to which pollinators improve public understanding of ecology.** Pollinators have been a flagship cause for ecosystem services research and are widely seen in press, charitable campaigns and even popular culture. To date however, no study has explored how this exposure has changed public understanding of ecology and ecosystem services. Understanding this could provide valuable insights into how to better communicate wider ecological issues and benefits (e.g. the importance of insects as natural pest control in agriculture) to the public at large and help generate support for broader environmental management initiatives.

5.  **How local knowledge can contribute to pollination management.**

# 3. Drivers of pollinator status & current and future risks

Authors: Deepa Senapathi, Adam J Vanbergen, Mark J F Brown

This section provides an update of the evidence (since Vanbergen *et al.* 2014) for direct drivers that are, or have the potential for, eliciting changes in the health or biodiversity of pollinators. Here we focus on <u>direct</u> drivers (proximate 'causes') that influence pollinators, as opposed to <u>indirect</u> drivers - such as human population or economic growth - that cause changes in the direct drivers and so ultimately influence pollinators. In this section, we assessed effects of land use [3.1] covering land-use change [3.1.1], urbanization [3.1.2]; conventional agricultural intensification [3.1.3]; agricultural GMOs [3.1.4]; pesticides (including insecticides, herbicides and fungicides) [3.2]; climate change [3.3]; invasive alien species [3.4]; and pests and diseases of bees [3.5].

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## 3.1 Land use - Deepa Senapathi

### 3.1.1 Land-use changes in habitat and landscape structure

Anthropogenic land-uses, including conventional intensive land management practices [3.1.3], often destroy, fragment or degrade semi-natural habitat in the landscape thereby simplifying the landscape (IPBES 2016b). This typically reduces or modifies the floral or nesting resources on which many pollinator species depend (Vanbergen *et al.* 2014; Clough *et al.* 2014; Baude *et al.* 2016). Such changes to semi-natural habitat resources are an important driver of pollinator population declines, species extinctions, or compositional changes in communities and plant-pollinator interactions (IPBES 2016b; Andersson *et al.* 2013; Ollerton *et al.* 2014; Marini *et al.* 2014; Aguirre-Gutierrez *et al.* 2015,).

Global assessments, meta-analyses, and specific studies from European countries (including Great Britain) show that increased landscape heterogeneity and resource-rich semi-natural habitat typically enhances richness and abundance of wild bees, wasps, flies and butterflies (Kennedy *et al.* 2013; Steckel *et al.* 2014; Aguirre-Gutierrez *et al.* 2015; Kleijn *et al.* 2015; Scheper *et al.* 2015; Senapathi *et al.* 2015). In addition, there is also evidence from Great Britain that landscape heterogeneity may buffer long-term declines in wild bee richness at local site levels (Senapathi *et al.* 2015). Recent evidence from studies in England show that providing floral resources enhanced bumble bee colony reproduction (Carvell *et al.* 2015), and that the proportion of high-value foraging habitat in the colony's vicinity (up to 1,000m) increased the survival of successive colony cycles and hence the bumble bee populations (Carvell *et al.* 2017).

Differences in species traits (e.g., feeding adaptations, mobility and foraging behaviour) govern the response of pollinator species to the effects of habitat changes and their ability to persist in poor-quality, structurally simplified, landscapes (IPBES 2016; Redhead *et al.* 2016). In landscapes with low resource diversity only those bee species able to forage over great distances in search of species-rich floral patches (Jha *et al.* 2013; Redhead *et al.* 2016) may ultimately persist. Overall, pollinator species with more specialised habitat or food requirements tend to be more vulnerable to land cover changes that alter the availability of food or nesting resources. Over time, this leads to the homogenization of pollinator communities in simplified landscapes dominated by common generalist species (Vanbergen *et al.* 2014; Marini *et al.* 2014; Weiner *et al.* 2014; Kleijn *et al.* 2015; IPBES 2016; Potts *et al.* 2016).

By altering pollinator foraging behaviour, through population-level effects on abundance, or differential species losses from an area, landscape-scale changes to habitat extent and attendant resources has the potential to alter reproductive success of plants dependent on insect pollination. Data on pollination deficits arising from landscape changes are generally scant although there are indications from a global meta-analysis (Aguilar *et al.* 2006) of a link between habitat fragmentation and decreasing plant reproductive success. In the UK, overall there remains little direct evidence of pollination deficits in wild plants, although marked declines in insect-pollinated wild plants are long-documented (Vanbergen *et al.* 2014; IPBES 2016). There are no UK studies showing crop pollination deficits explicitly linked to landscape changes in habitat resources. Indeed some international studies show that abundant generalist bee species able to persist in contemporary landscapes deliver the majority of crop pollination services (Kleijn *et al.* 2015, Winfree *et al.* 2015). Yet because

pollinator species have differential responses to the same environmental perturbations (response diversity), maintaining diverse pollinator communities with complementary traits or behaviours can increase the stability of crop pollination over space, time and in the face of environmental change (IPBES 2016; Potts *et al.* 2016; Senapathi *et al.* 2017).

Aside from direct effects of landscape characteristics on pollinator biodiversity, there remains scope for future changes through interaction with other drivers (Vanbergen and the Insect Pollinators Initiative 2013; Senapathi *et al.* 2017). For example, Park *et al.* (2015), in the USA, showed how the effects of pesticides on bee abundance and species richness (see Section 3.2) could be ameliorated by surrounding semi-natural habitat. Alterations to bee species distributions due to climate changes raises the risk of an interaction with the landscape configuration of habitats, posing challenges to pollinators (Vanbergen and the Insect Pollinators Initiative 2013; Kerr *et al.* 2015; IPBES 2016).

## 3.1.2 Urbanization

Urban land cover is increasing globally, and in the UK, and the resultant habitat loss and fragmentation is an important driver of plant–pollinator interactions (Harrison & Winfree 2015, Jha & Kremen 2013; IPBES 2016). Early evidence indicated that pollinator abundance and richness tends to decrease with increasing urbanization (e.g. Ahrne *et al.* 2009; Bates *et al.* 2011). Some recent evidence from England comparing pollinator communities in urban and non-urban landscapes revealed that towns and cities can support higher species richness of bees compared to agricultural land, but no difference in bee richness for urban sites compared to nature reserves (Baldock *et al.* 2015). There is also some evidence that urban bee communities provide a benefit by pollinating urban crops and garden plants (Lowenstein *et al.* 2015; Theodorou *et al.* 2016; but see Theodorou *et al.* 2017 for contrary evidence). Evidence from Belgium, France and Great Britain shows that comparatively bumble bees appear less adversely affected by urban development than other pollinator taxa such as hoverflies (Verboven *et al.* 2014; Baldock *et al.* 2015; Deguines *et al.* 2016), and a recent study in SE England showed higher reproductive output by bumble bee colonies in urban areas relative to agricultural areas (Samuelson *et al.* 2018). However, analogous to the situation in contemporary simplified landscapes dominated by convential intensive agriculture, urbanization tends to homogenise bee and the overall communities with generalist species predominating (Baldock *et al.* 2015; De Palma *et al.* 2015; Deguines *et al.* 2016; IPBES 2016). There is also a need to research how environmental pollution in urban areas affects bee foraging and health, there is an example of diesel pollution interfering in bee capacity to locate flowers during foraging (Girling *et al.* 2013). In general, urbanization effects on non-bee pollinators has been under-researched and is complex Effects depend on the type of land that is converted and the extent of management of different urban green spaces (gardens, amenities, railway embankments), which can benefit pollinators. Further information is required to augment specific urban habitat management (e.g. restoration of brown-field sites) to benefit pollinators and flowering plants (Tarrant *et al.* 2013).

## 3.1.3 Conventional agricultural intensification

Conventional agricultural intensification is defined as the industrial management of large-scale monocultures with high levels of agrichemical inputs, tillage, grazing, mowing and mechanization (Kovács-Hostyánszki *et al.* 2017). Such farm management practices, by reducing both foraging (flowering plants) and nesting (undisturbed soil or vegetation) resources for pollinators, decrease pollinator diversity dramatically and influence the effectiveness of pollination services (IPBES 2016).

Monocultures of mass flowering crops (e.g. orchards, oilseed rape) do provide nectar and pollen sources that support wild bee populations (Riedinger *et al.* 2016; Holzschuh *et al.* 2016). This transient pulse of food, however, does not compensate for the losses of other food sources like wildflowers that sustain most pollinators throughout their life cycle (Potts *et al.* 2016, Baude *et al.* 2016; IPBES 2016). Moreover, large-scale cultivation of oilseed rape may even disrupt pollination services in the wider landscape (Holzschuh *et al.* 2016).

Evidence from across Europe and North America shows the proportion and management intensity of agricultural land in landscapes lowers pollinator abundance and diversity (Marini *et al.* 2014; Steckel *et al.* 2014; Connelly *et al.* 2015; Scheper *et al.* 2015). Although the magnitude varies with the type of agriculture or crop, and the management regime (Le Feon *et al.* 2013; Brittain *et al.* 2010; Marini *et al.* 2014). There is evidence that increasing the diversity of crop types on the farm, maintaining ecological infrastructure or adopting 'ecologically-intensive' management practices commensurate with maintaining agricultural production alongside biodiversity conservation and managing ecosystem services, can ameliorate many of the adverse impacts of conventional intensive management (IPBES 2016; Potts *et al.* 2016; Kovács-Hostyánszki *et al.* 2017).

Bee functional diversity can be lower in agricultural landscapes compared to natural habitats. For example, in a recent American study, Forrest *et al.* (2015) found that bee assemblages in farmland had low functional diversity compared to those in nearby natural communities. De Palma *et al.* (2015) collated data for 257 bee species from multiple studies across Europe and found that smaller-bodied species and those with shorter flight seasons were less likely to be present in areas of intensive agriculture. Such filtering and removal of species and their traits from agricultural pollinator communities presents a risk to the sustained pollination function as the environment changes. There is some evidence, however, to suggest that within intensive farmed areas, existing linear landscape features can act as functional biological corridors facilitating pollinator movements (Van Geert *et al.* 2010) and restoring flowering hedgerow can promote bee populations and diversity (M'Gonigle *et al.* 2015). Evidence is also growing that by increasing wildflower patch sizes and providing high-value floral resources in the local agricultural landscape can lead to increases in wild bee density (Bennett & Isaacs 2014) and promotion of bumble bee colony reproduction and population persistence (Carvell *et al.* 2015; Carvell *et al.* 2017).

## 3.1.4 Agricultural genetically modified organisms (GMO)

A particular form of intensive agricultural management is the use of crops that are genetically modified organisms (GMOs) possessing transgenes that confer herbicide tolerance (HT) or insect resistance (IR). Currently in the UK (and most of the EU) there are no agricultural genetically modified organisms (GMOs) grown commercially[1]. Consequently the possibility that GMOs can drive changes in pollinators or pollination in the UK/EU is at present very low. A potential risk to migratory pollinators (e.g. certain butterfly species) coming from GMO cultivation beyond the EU could be hypothesised but there is no information to support that at present. Should GM-crop cultivation become common practice in the future there will need to be consideration of the opportunities to benefit pollinators (i.e. reduction in insecticide pressure in crop fields) versus the risks to populations and diversity (e.g. no relaxation of insecticide use due to pest resistance; loss of floral resources due to HT-crop management) coming from GM-crop management. Consequently, we can only infer the risks and opportunities for pollinators from the cultivation of GM crops from global evidence (IPBES, 2016) alongside the UK farm-scale evaluation carried out between 1999 and 2006[2], although the principal focus of the latter was not on risks to insect pollinators and pollination. The current global evidence basis on direct and indirect effects of GM-crops on pollinators is generally limited compared to other drivers, but many of the indirect effects (management related) can be inferred from the known impacts of conventional agricultural intensification (IPBES, 2016; Kovács-Hostyánszki *et al.* 2017).

### 3.1.4.1 Insect-resistant (IR) crops

Insect-resistant (IR) crops, such as those producing *Bacillus thuringiensis* (Bt) toxins, are engineered to target specific crop pests, most often larvae of lepidopteran (butterflies/moths) or coleopteran (beetle) herbivores (IPBES, 2016; Potts *et al.* 2016). There have been no reports of direct lethal effects (toxicity) of insect-resistant (IR) crops on honey bees or other Hymenoptera; while there have been few studies assessing sub-lethal effects on bees or other pollinators and the evidence from these is inconclusive (IPBES, 2016). There is a general scarcity of data on effects on other pollinator groups (e.g., hoverflies), although there have been some records

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[1]https://www.gov.uk/food-safety-as-a-food-distributor/genetically-modified-foods
[2]http://webarchive.nationalarchives.gov.uk/20080306073937/http://www.defra.gov.uk/environment/gm/fse/

of lethal effects on butterfly species (IPBES, 2016). Overall the direct risk to pollinators from IR crops is likely to be reduced when the pollinator species is not closely related phylogenetically to the target pest organism (Potts *et al.* 2016).

A potential indirect benefit to pollinators from cultivation of insect resistant (IR) crops could arise where there is a concomitant reduction in insecticide use, which if sustained could reduce pressure on non-target insects like pollinators (IPBES, 2016; Potts *et al.* 2016). However, this reduction in insecticide use varies with crop species and geographic region according to the prevalence of various pests and the secondary outbreaks of non-target pests or primary pest resistance often lead to resumption of the pesticide regime (IPBES, 2016). The effects of insect-resistant (IR) crop use and associated reductions in pesticide use on pollinator abundance and diversity remain to be determined.

### 3.1.4.2 Herbicide tolerant (HT) crops

The management of HT crops requires the regular application on herbicides that will lead to reduced weed populations. Based on what is known about the relationship between wild plant populations and pollinator foraging, nutrition and densities, this management is highly likely to diminish the supply and diversity of pollen and nectar sources available to pollinators in the farmed landscape (Baude *et al.* 2016; Clough *et al.* 2014; Dicks *et al.* 2015; IPBES, 2016). The limited evidence from the few available studies suggest that the effects of HT-crop management can lead to a reduction of pollinators in fields (Bohan *et al.* 2005; Haughton *et al.* 2003), but the long-term consequences for pollinator abundance and diversity of foraging in herbicide tolerant (HT)-crop fields is unknown (IPBES, 2016).

There is some evidence of transgene escape from GM-crops and persistence in non-GM crops or wild plants through hybridisation or introgression. The ecological and evolutionary consequences of this for pollinators and pollination remain to be investigated thoroughly (IPBES, 2016; Potts *et al.* 2016).

### 3.1.5 Knowledge gaps

- The precise spatio-temporal configuration of habitats in local landscapes to support pollinator populations over time.

- The level of habitat diversity, heterogeneity or configuration required for sustained delivery of pollination services to crops and/or wild plants.

- Quantification of the impact of landscape level habitat changes on non-bee taxa, particularly flies.

- The habitat requirements and relative roles of pollinator taxa in wild plant/crop pollination, to understand delivery vs resilience of pollination function.

- The impact of chemicals in the urban and suburban environment (pesticides, artificial fertilisers, atmospheric pollution) on pollinator health.

- Investigation of multiple land-use drivers and their interplay in affecting pollinator biodiversity.

- The efficacy of habitat restoration in degraded urban and agricultural settings for attracting and sustaining wild pollinators.

- Assessment of the sub-lethal direct risks of Bt-toxins (e.g. expression/secretion into pollen/nectar) to different pollinator taxa (social and solitary bees, hoverflies).

- Cost-benefit analyses of GM-crop management for pollinators, pollination and pest control.

## 3.2 Pesticides, herbicides, and fungicides – Mark Brown

Chemicals are used broadly across agricultural, rural, suburban, and urban landscapes to control pests and weeds. However, they may have a range of unintended direct and indirect effects on both wild and managed pollinators (Vanbergen *et al.* 2014). For example, a recent meta-analysis confirmed that insecticides had generally negative effects on learning and memory in bees when applied at field-realistic dosages (Siviter *et al.* 2018). Considerable attention has been focused recently on the potential impacts of a class of insecticides called neonicotinoids, but other pesticides, herbicides, and fungicides also have the potential to threaten wild and managed pollinators. Here we focus largely on neonicotinoids, as this is where most progress has been made in terms of new knowledge and understanding since Vanbergen *et al.* (2014).

### 3.2.1 Insecticides

Neonicotinoids are the most widely used class of insecticides worldwide (Simon-Delso *et al.* 2015; IPBES 2016). Despite their approval by numerous regulatory authorities, many scientific studies have demonstrated significant negative impacts on bumble bees and honey bees (reviewed in Godfray *et al.* 2014, 2015; van der Sluijs *et al.* 2015; IPBES 2016). In 2013, three neonicotinoids – clothianidin, imidacloprid, and thiamethoxam – were placed under a limited moratorium within the EU (including England). Following this, the UK Expert Committee on Pesticides reviewed available evidence in 2017, concluded it supported the 2013 restrictions and noted concerns about usage outside of completely enclosed systems. In 2018, the European Commission completely prohibited the outdoor use of these compounds, following an EFSA risk assessment (2018)[3] on the threat to bees from these three neonicotinoids, which concluded that risks to bee health from use of these compounds were high outside of permanent greenhouse structures (EFSA 2018a,b,c).

Since publication of Vanbergen *et al.* 2014 in support of the NPS, there have been many studies worldwide on the risk to pollinators from neonicotinoids; below we detail a few examples. The IPBES 2016 global assessment of pollinators and pollination assessed the global evidence and concluded that there was evidence that neonicotinoid insecticides incurred lethal and sub-lethal effects on bees and some evidence of impacts on the pollination they provide. Recent farm-scale experiments in Europe (including England) have demonstrated significant negative impacts of neonicotinoids on bumble bee and solitary bee survival and performance at actual field exposure (Rundlöf *et al.* 2015, Woodcock *et al.* 2017). Analysis relating 18 years of UK national wild bee distribution data for 62 species to amounts of neonicotinoid use in oilseed rape revealed significant correlations with solitary bee population declines (Woodcock *et al.* 2016). Laboratory and semi-field experiments have produced an array of results including: impacts on colony founding and population persistence (shown through modelling) in bumble bees (Baron *et al.* 2017a, Baron *et al.* 2017b); evidence that bumble bees actively prefer neonicotinoid contaminated food (Kessler *et al.* 2015); adverse impacts on foraging behaviour and effectiveness but equivocal effects on bumble bee colony health (Stanley *et al.* 2017); and even possible direct impacts of neonicotinoid exposure on pollination services (Stanley *et al.* 2015).

The impacts of neonicotinoid use on honey bee health remain unclear, with studies finding no effect, negative effects, and even positive effects (Tsvetkov *et al.* 2017, Woodcock *et al.* 2017,EFSA 2018a,b,c); such variation has been suggested to be driven by factors such as forage availability and stressors such as pathogens (Woodcock *et al.* 2017). Adverse effects on inter-annual colony reproduction may emerge either when the bees depend greatly on a treated crop (nutritional deficits) or there are other stressors (e.g. disease) present (Vanbergen and the Insect Pollinators Initiative 2013; Woodcock *et al.* 2017). There are comparatively few studies of neonicotinoid impacts on non-bee taxa (IPBES 2016), but at least one shows no impact on a hoverfly species (Basley *et al.* 2018).

Studies have found contamination of wild plants by neonicotinoids near seed-treated crops (Botias *et al.* 2016), as well as in plants sold for garden use (Lentola *et al.* 2017), although explicit links to bee health have not been shown. Analysis of honey demonstrated widespread, global exposure of honey bees to neonicotinoids (Mitchell *et al.* 2017), while in the UK neonicotinoids were still detectable in more than 20% of 109 honey samples collected the year after the EU moratorium was put in place (Woodcock *et al.* 2018); potential mechanisms

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[3] http://www.efsa.europa.eu/en/supporting/pub/en-1378

behind this include (i) the uptake of neonicotinoids from the soil by oil seed rape crops planted in rotation after seed-treated cereal crops, and (ii) the movement of honey made in the previous year within the hive.

Given the current and potential future restrictions on neonicotinoids, a new class of insecticides – sulfoximines – have been developed and are approved for use in some jurisdictions whilst still going through regulatory approval in others (Brown *et al.* 2016). A recent semi-field study demonstrated significant negative chronic and sub-lethal impacts of sulfoxaflor, a sulfoximine, on bumble bee colony growth and reproductive success (Siviter *et al.* 2018b), but our knowledge is incomplete and further studies are needed.

## 3.2.2 Other agrochemicals

While the current focus has been on neonicotinoid insecticides (see above), a broad range of pesticides, fungicides, and herbicides are used in ways that can result in exposure to pollinators (Mullin *et al.* 2010, Vanbergen *et al.* 2014, Botias *et al.* 2017). However, the potential impact of such agrochemicals, whether used correctly or not, on pollinator health remains unclear. Herbicides, which account for nearly 40% of total pesticide use worldwide, may reduce the availability of forage resources, but how this relates to pollinator abundance or diversity remains unclear (Bohnenblust *et al.* 2016)(see section 3.1.4.2). Fungicides are one of the most abundant biocides in wild bumble bees (Botías *et al.* 2017) and in honey bee hive material (Mullin *et al.* 2010, Daniele *et al.* 2018, Tosi *et al.* 2018). Low doses of fungicide can kill honey bee mid-gut cells, resulting in proliferation of the fungal disease *Nosema ceranae* (Huang *et al.* 2013), as well as altering the honey bee microbiome (Mao *et al.* 2017). In honey bees, co-exposure with fungicides increases the toxicity of neonicotinoids (Tsvetkov *et al.* 2017). At the landscape-scale, patterns in fungicide use predicted pathogen prevalence and range contractions in US bumble bees (McArt *et al.* 2018). Finally, at the farm-scale, many agrochemicals are applied as part of formulations that include adjuvants that enhance the activity of the relevant active ingredient. However, these adjuvants not only enhance toxicity of these formulations (Mullin 2015), but may also have toxic effects in their own right (Chen *et al.* 2018).

## 3.2.3 Knowledge gaps

- The cumulative effect of total use of herbicides, fungicides and insecticides within a growing season on the health, survival and performance of pollinators.
- Develop new integrated methods of monitoring changes in pollinator exposure that may provide early warning of increasing risk.
- Quantification of the effectiveness of the EU moratorium and ban in reducing exposure to neonicotinoids as would be predicted from half-life predictions, and monitor pollinator population response in the absence of neonicotinoids.
- The impact of pesticides and their combinations on non-bee pollinators.
- Understanding the relative impact of pesticides on pollinator populations and pollination services compared to other drivers.
- Determining the impacts of herbicides on floral availability and pollinator nutrition.
- Determining the impacts of fungicides on pollinator health across scales, from individuals to landscapes.
- Understanding how farmers will respond, in terms of crop management and agrochemical use, to the permanent restriction on neonicotinoid use, and the likely consequences of these changes for pollinators.
- The potential impacts of sulfoximines (and other emerging agrochemicals) on pollinators, and how they can these be assessed within the framework of risk assessment prior to their approval for use.
- Determining the toxicity of adjuvants, and how they enhance toxicity of other agrochemicals for pollinators.

## 3.3 Climate change – Deepa Senapathi

Climate change is anticipated to bring about changes in rainfall distribution, wind patterns, temperature, air pollution and occurrence of extreme weather events, among other environmental changes (IPCC 2014); for some species of pollinator, such as bumble bees and butterflies climate change has already resulted in, and is projected to continue to change ranges, alter abundances, and shift the seasonal activity patterns of pollinators. An increasing number of observational and experimental studies across many organisms provide strong evidence that climate change has contributed to the overall spring advancement observed especially in the Northern Hemisphere (Settele *et al.* 2014), leading to potential mismatches in plant-pollinator phenology, thereby de-synchronising ecological interactions and threatening ecosystem function (Thackeray *et al.* 2016). Plants reproduction affected by climatic events such as frost have also been shown to change floral traits and negatively impact pollinator visitation (Pardee *et al.* 2018).

Evidence suggests that climate has a significant impact on plant-pollinator interactions, even within relatively narrow latitudinal gradients (Petanidou *et al.* 2018), and where plant-pollinator interactions have been studied in combination, emerging evidence indicates that mismatches in the timing of effective interactions may result in reduced performance at both trophic levels (Kudo & Ida 2013). While long-term studies are rare due to lack of historic data, experimental studies such as Schenk *et al.* (2018), on *Osmia bicornis* show that even short temporal mismatches can cause clear fitness losses in solitary bees.

While the main focus of climate driven mismatches has alluded to spring climate and phenology, published work from the UK, clearly shows that autumn/winter temperature shifts can drastically alter the diapause or overwintering behaviour in insects (Coleman *et al.* 2014), and can even prevent emergence altogether (Bale & Hayward 2010).  Quantitative data on key pollinators is severely lacking, however experimental studies on *Bombus terrestris*  show that diapause is already being disrupted with queens attempting to establish winter colonies (Stelzer *et al.* 2010) and weight loss, survival, and emergence times being impacted in the mason bee *Osmia cornuta* (Bosch & Kemp 2004). Additionally, there is some evidence from the USA that daily activity patterns may change with climate change (e.g. Rader *et al.* 2013).

There is substantial evidence that terrestrial plant and animal species' ranges have moved in response to warming over the last several decades and this movement will accelerate over the coming decades (Settele *et al.* 2014). A synthesis of range shifts (Chen *et al.* 2011) indicates that terrestrial species (e.g. butterflies) have moved poleward about 11-17 km per decade across various sites including Europe and North America. Kerr *et al.*'s (2015) study on bumble bees across Europe and North America (using over 110 years of observed data) found cross-continentally consistent trends in failures to track warming through time at species' northern range limits, range losses from southern range limits, and shifts to higher elevations among southern species. There is however, tremendous variation in range shifts among species and species groups.

The current climatic niche for 56 European bumble species have been mapped and projected future climatically suitable conditions using three climate change scenarios for the years 2050 and 2100 presented in a risk atlas (Rasmont *et al.* 2015). While under a moderate change scenario only three species are projected to be at the verge of extinction by 2100, 14 species are at high risk under an intermediate change scenario. Under a most severe change scenario as many as 25 species are projected to lose almost all of their climatically suitable area, while a total of 53 species would lose the main part of their suitable area.

The attribution of extinctions to climate change is much more difficult, but there is a growing consensus that the interaction of other global change pressures with climate change poses the greatest threat to species (Gonzalez-Varo *et al.* 2013; Vanbergen and the Insect Pollinators Initiative 2013). While there is no scientific consensus concerning the magnitude of direct impact of climate change on extinction risk, in the context of other global change pressures this will contribute substantially to increased extinction risks over the coming century (Settele *et al.* 2014). A recent modelling study on hoverfly species in Europe has shown how species adapted to high mountains are projected to almost vanish from the Balkans and only regions of the Alps would remain suitable for them (Radenkovic *et al.* 2017). Another modelling study on butterflies in Britain has showed that while widespread drought-sensitive butterfly population extinctions could occur as early as 2050, measures to manage landscapes for pollinators and in particular to reduce habitat fragmentation could significantly improve the probability of persistence (Oliver *et al.* 2015). While regional studies show

variation in species responses, a global study by Urban (2015) suggests that extinction risks will accelerate with future global temperatures, threatening up to one in six species under current policies with extinction risks highest in South America, Australia, and New Zealand.

### 3.3.1 Knowledge gaps

- Experimental studies of climate change impacts on insect pollinators have mainly focused on temperature while the impact of changes in other variables such as precipitation and $CO_2$ are less well characterised.

- Most of the evidence of phenological mismatches consider the effect of spring temperatures; but a comprehensive assessment of responses to climate warming across other seasons as well as the impact of extreme events is lacking.

- While models are able to project likely spatial and temporal shifts in pollinators, the impacts of climate change, and partially extreme weather events, on pollination services are almost entirely unknown.

- There is as yet insufficient evidence to determine whether climate driven disruptions of plant-pollinator interactions are linear or if there may be non-linear responses (e.g. tipping points).

## 3.4 Invasive alien species - Adam Vanbergen

Invasive alien (non-native) species[4] spread as a consequence of the global growth in economic wealth, trade, commerce, and transport efficiency (Dawson *et al.* 2017; Hulme, 2009; Mack *et al.* 2000). Invasive alien species of plants can create risks and opportunities for pollinator nutrition and re-organise species interactions to affect native pollination and community stability (IPBES, 2016; Vanbergen *et al.* 2018). Effects of invasive alien species on native systems tend to be complex and contingent on the abundance and eco-evolutionary function of the invader and the evolutionary history of the recipient ecosystem (IPBES, 2016). For example, invasive alien plant species with certain ecological traits are readily incorporated into interaction networks, where, once abundant, they can substantially modify native pollination systems. Whereas the potential risk from invasive alien predators is from direct consumption lowering population sizes (IPBES, 2016; Vanbergen *et al.* 2018). In the following summary, we assess the available evidence for impacts on pollinators and pollination from invasive alien species of flowering plants and predators, as the two functional groups currently representing a threat in the UK and Europe. The risk from pests and pathogens, some of which are introduced aliens, are dealt with elsewhere in this update (section 3.5).

### 3.4.1 Alien flowering plants.

Most of the following research on alien flowing plants comes from outside the UK, although some of the reviews and meta-analyses include UK studies. Abundant invasive alien plant species possessing enduring floral displays that offer copious nectar and pollen rewards provide a substantial food resource attracting and dominating the diet of pollinators (Bartomeus *et al.* 2010; Dietzsch *et al.* 2011; Kleijn & Raemakers, 2008; Montero-Castaño & Vilà, 2017; Pysek *et al.* 2011; Stiers & Triest, 2017; Vila *et al.* 2009). As well as the nutritional opportunity plant invasions can present to pollinators there may also be hitherto unstudied risks for pollinator health. One potential risk is that invasive alien plant species may alter the nutritional quality of landscapes for pollinators. Experiments show that pollinators have specific requirements for a balance of essential nutrients (e.g. ratios of protein: lipid: carbohydrates) (Sedivy *et al.* 2011; Stabler *et al.* 2015; Vaudo *et al.* 2016). If a plant invasion leads to reductions in the availability of particular essential nutrients provided by diverse floral resources, then specific nutritional deficits for pollinators might arise (Vanbergen *et al.* 2018). Secondary plant

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[4] The Convention on Biological Diversity (www.cbd.int/invasive/WhatareIAS.shtml) defines invasive alien species (IAS) as those intentionally or accidentally introduced beyond natural ranges by human actions, which subsequently spread as vigorously growing populations that impact on biota, ecosystems, and society.

defence compounds in alien pollen and nectar can be differentially toxic to native pollinator species (Arnold *et al.* 2014; Praz *et al.* 2008; Tiedeken *et al.* 2016); so although requiring further investigation the novel biochemistry of invasive alien plants where they dominate pollinator diets might represent another potential risk. Dominance of plant communities by invasive alien species could also modify community-wide flowering phenology to affect the period over which floral resources are available to foraging insects, with implications for reproductive success (Stelzer *et al.* 2010; Vanbergen *et al.* 2018).

These risks to pollinators from changes to floral resource quality or phenology could impact on pollinator survival, health and reproduction. However, there are comparatively few recorded accounts of alien plant invasions in the UK (or comparable regions elsewhere) consistently lowering pollinator diversity or abundance (Lopezaraiza-Mikel *et al.* 2007; Moron *et al.* 2009; Nienhuis *et al.* 2009). This suggests that either pollinators can compensate physiologically or behaviourally for changes in nutrient quality or availability associated with alien species (Harmon-Threatt & Kremen, 2015); or that effects are subtle, chronic (possibly undetected) or only adversely affect pollinators in combination with other stressors (Schweiger *et al.* 2010; Vanbergen & the Insect Pollinators Initiative, 2013).

Although direct links between invasive alien plants and pollinator decline are either absent or hard to prove, there is abundant evidence that they substantially modify plant-pollinator interaction networks. Once the alien plant species is established, the flexible foraging behaviours and generalised diets of many pollinators mean alien plants are readily integrated into and come to dominate native plant-pollinator networks (Albrecht *et al.* 2014; Bartomeus *et al.* 2008; Goodell & Parker, 2017). Following integration, the invasive plant often realises a key structural or functional role in the stability of plant-pollinator networks. Where the alien plant operates as a key hub of species interactions it can provide cohesion to the network potentially lowering the chance of subsequent secondary extinctions, as seen with simulations of networks modified by livestock grazing (Vanbergen *et al.* 2017). Alternatively, should the native pollinators and plants become overly dependent on the invader, then the removal of the invasive (by a second environmental change or management) might precipitate secondary extinction cascades that threaten the local pollination system (Vanbergen *et al.* 2018). However, this risk may be lessened by the generalised and highly flexible foraging behaviour of many pollinators, which means that interaction networks are dynamic and can rewire and reform to maintain functionality, albeit in different ways.

The main risk from invasive alien plants comes from their domination of interactions, which has consequences for the insect pollination of co-flowering native plants. Overall, alien plants tend to elevate local pollinator activity. Many studies point to this decreasing visitation rates to co-flowering native plants, which indicates that competition with invasive plants for pollination services prevails (Brown *et al.* 2002; Dietzsch *et al.* 2011; Morales & Traveset, 2009; Thijs *et al.* 2012). Although under some situations there is evidence they can instead facilitate native pollination suggesting a spill-over benefit for co-flowering native plants of the alien-induced uplift in pollinator activity (Albrecht *et al.* 2016; Carvalheiro *et al.* 2008; Masters & Emery, 2015; McKinney & Goodell, 2011; Russo *et al.* 2016). Whether local facilitation or competition pre-dominates is likely to depend on the relative abundance and distribution of the alien plant, the level of floral rewards it offers c.f. native species, and the composition of species and traits in the recipient habitat (Albrecht *et al.* 2016; Carvalheiro *et al.* 2014; Montero-Castano & Vila, 2012; Russo *et al.* 2016; Stout & Casey, 2014).

Invasive alien plants through their interactions with pollinators can disrupt native plant reproduction by affecting pollen flows. In dominating interaction networks, the alien plant may cause improper pollen transfer either by reducing the transfer of native (conspecific) pollen or by insect deposition of alien (heterospecific) pollen leading to reproductive interference (stigma clogging or chemical inhibition of pollen germination). Both mechanisms may reduce native plant reproduction (Brown *et al.* 2002; Bruckman & Campbell, 2016b; Chittka & Schurkens, 2001; Thijs *et al.* 2012); yet the extent to which this occurs is complicated by plant compensatory mechanisms, e.g. self-pollination, that can assure reproduction (Dietzsch *et al.* 2011; Emer *et al.* 2015; Morales & Traveset, 2009).

Overall, impacts on native plant pollination and reproduction tend to be greater where the alien plant produces greater densities of highly-rewarding flowers relative to the native flora, where alien and native are phylogenetically related or possess similar phenology/structure of floral displays (Bruckman & Campbell, 2016a; Luísa Gigante Carvalheiro *et al.* 2014; Herron-Sweet *et al.* 2016; Pysek *et al.* 2011). Where some or all

of these conditions are met then pollinator sharing between the native and the invasive plant species rise to the point where there is an impact, positive or negative, on the native plant (e.g. Thijs *et al.* 2012).

### 3.4.2 Alien invasive predators.

Where predators are introduced beyond their native range they typically exert strong top-down pressure on plant pollination and fitness by consumption of pollinators (IPBES, 2016). The accidental introduction (2004) from Asia into France and subsequent spread across the western European region, of the predatory Asian hornet (*Vespa velutina*) represents a new threat to honey bee populations (Budge *et al.* 2017; Keeling *et al.* 2017; Monceau *et al.* 2014), already stressed by other pressures (IPBES, 2016; Potts *et al.* 2016; Vanbergen & the Insect Pollinators Initiative, 2013). The long term effects of the invasion by the hornet and its attacks on honey bees remain to be determined. However, once the hornet becomes established, nest densities are high (Bertolino *et al.* 2016; Franklin *et al.* 2017; Monceau & Thiery, 2017) and the western honey bee *Apis mellifera* does not possess the evolved anti-predator behaviour of the Asian honey bee (*A. cerana*) (Tan *et al.* 2007; Tan *et al.* 2012), thereby increasing their potential vulnerability. Furthermore, effects of invasive alien predators on the food web affecting native pollination can be highly complex. For instance, there are examples elsewhere in the world where invasive alien insect predators (*Vespula* wasps) aggressively and directly compete with pollinators for nectar or other food sources, sometimes with consequences for wild pollinators and plant reproduction (Hanna *et al.* 2013, 2014; Markwell *et al.* 1993). The Asian horn*et al*so attacks wild pollinators and is a floral visitor so there are potential risks to the network of interactions affecting pollinator communities and pollination processes.

Generally, in the UK context the impact of invasive alien species on pollinators and pollination can be considered, on the basis of the current evidence, to be less profound than other pressures (e.g. land management, pests and pathogens) (Vanbergen *et al.* 2014). It is possible, however, that some risks may have been overlooked or understudied (Vanbergen *et al.* 2018). Moreover, as the climate changes the level of pressure from invasive alien species on pollinators may rise as new niches are created and native pollinators and plants come under direct climatic or other stresses (IPBES, 2016; Schweiger *et al.* 2010; Vanbergen & the Insect Pollinators Initiative, 2013).

### 3.4.3   Knowledge gaps

- The nutritional costs and benefits (nutrient ratios, novel plant toxins) to pollinators from alien plants that dominate plant-pollinator networks.

- How plant species invasion alters community-wide flowering phenology to influence the temporal dynamics, composition, and diversity of pollinator communities and pollination services.

- Forecasting the risk of emerging invasive species (e.g. Asian hornet) for managed and wild pollinators.

- The consequences of species invasions (plants, animals or pathogens) for the structure and stability of holistic networks that comprise a range of functional biodiversity (mutualists, decomposers, herbivores etc.).

- Understand the mechanism (temporal dynamics, competition, mutualism dependence) underpinning re-organisation of pollinator network structure by invasive species.

- Identify eco-evolutionary constraints, e.g. invader genetic diversity or effective population size, governing invasion and evolutionary trajectories post-invasion to predict future risk.

- Establish the role of co-flowering native plant species, through their influence on foraging behaviour of different pollinator groups, in aiding alien plant establishment and dynamics.

- If there is an effect on insect-dependent crop production of changes in pollinator foraging due to invasive alien plants.

- The risk from global horticultural and agricultural trade for the spread of alien plant pathogens that can modify plant physiology to affect pollinator interactions and plant reproduction.

## 3.5 Pests and diseases of bees – Adam Vanbergen

Parasites and pathogens can play a major role in the dynamics of their host populations.  Endemic, epidemic, and emerging infectious diseases have all been linked to impacts on pollinator health (Vanbergen *et al.* 2014). Numerous studies within the UK and across the globe have examined the impacts of parasites and pathogens on honey bee health, but our knowledge of their impacts on other pollinators is limited. The increasing use of molecular approaches is dramatically enhancing our understanding of how the landscape of parasites and pathogens is changing for pollinators.

### 3.5.1 Impacts on honey bees

Vanbergen *et al.* (2014) identified numerous pests, pathogens and diseases of managed honey bees. Of these, only a subset appears to pose threats to managed populations in England. European (*Melissococcus plutonius*; EFB) and American (*Paenibacillus larvae*; AFB) foul broods remain at low levels (<2% and <1%, respectively, with no recent (2013-2018) changes in incidence; (http://www.nationalbeeunit.com/) in managed honey bees in the UK, including England. These notifiable diseases can cause colony death, although their presence at such low levels, as a result of effective national monitoring and control, means that they currently do not pose a major threat to honey bee health in the UK. In contrast, the invasive mite *Varroa destructor* is ubiquitous across England and, combined with the co-occurring *Deformed wing virus* (Wilfert *et al.* 2016), poses a major threat to honey bee health. Recent work suggests that a new, more virulent strain of *Deformed wing virus*, linked to overwintering honey bee mortality, is spreading throughout honey bee populations in Europe, including the UK (McMahon *et al.* 2016; Natsopolou *et al.* 2017). The threat of *Varroa destructor* and its associated *Deformed wing virus* is managed in UK honey bee populations by bee husbandry techniques and chemical varroacides. It is unclear how either *V. destructor* or *Deformed wing virus* are linked to declines in managed honey bees in the UK. However, a recent cross-European study that included England and Wales found a winter colony loss of ~25% across 2012-2013, and a subsequent summer colony loss in 2013 of ~8% (Jacques *et al.* 2017). The major predictor of these losses was beekeeper background and apicultural practices, resulting in high levels of *V. destructor* infestation (Jacques *et al.* 2017), indicating the importance of appropriate management of this pest.

A number of other pests and diseases pose potential threats to the health of honey bees in England, including the Asian hornet (*Vespa velutina*, Monceau *et al.* 2014), the small hive beetle (*Aethina tumida*, Elzen *et al.* 1999), and Tropilaelaps (*Tropilaelaps clareae* and *T. mercedesae*, DEFRA 2005). The latter two (small hive beetle and Tropilaelaps) are statutorily notifiable under EU legislation, meaning that action must be taken to prevent arrival of these pests into the EU and to stop spread should they arrive. However, only the Asian hornet has so far been sighted in England (in 2016; Keeling *et al.* 2017). Models suggest that, without control, this invasive predator of honey bees could achieve high population density at national levels within 10 years, indicating the importance of current management and control strategies (Keeling *et al.* 2017). For more information on the potential impacts of the Asian hornet on managed honey bees, see section 3.4.2 above.

As is clear from the above, the work of the National Bee Unit (http://www.nationalbeeunit.com/) and local and national beekeeping organisations is central to controlling the threat and impact of pests and diseases in managed honey bees in England. See section 4.3.1 for details of the effectivity of management practices for honey bees.

## 3.5.2 Impacts on wild bumble bees

Bumble bees are known to host a range of endemic pathogens and parasites, many of which cause mortality and may control host population dynamics (Schmid-Hempel 1998; Antonovics & Edwards 2011; Vanbergen *et al.* 2014), but none of these have been causally linked to declines in wild bumble bee populations in the UK. Recent work has demonstrated that wild bumble bees share viral and fungal pathogens with managed honey bees in the UK (Fürst *et al.* 2014; McMahon *et al.* 2015), and that these pathogens can negatively impact bumble bees under laboratory conditions (Graystock *et al.* 2013a; Fürst *et al.* 2014). However, the directionality and intensity of cross-species transmission remains unknown, as well as whether these pathogens have any impact on wild bumble bee populations (Fürst *et al.* 2014). Commercial bumble bees imported into the UK for pollination may act as a source of parasites and pathogens for wild populations (Graystock *et al.* 2013b). However, the introduction of new pathogen screening approaches in commercial bumble bee production facilities may have reduced or eliminated this risk (Graystock *et al.* 2016). Transmission of parasites and disease from commercial to wild bumble bees has been shown or suggested at various sites around the globe (Meeus *et al.* 2011; Graystock *et al.* 2016). Whether transmission of pathogens and parasites from commercial to wild bumble bees occurs in the UK, and, if so, whether there are concomitant impacts on wild bumble bee populations remains unknown.

## 3.5.3 Impacts on wild solitary bees and other pollinators

Knowledge of pathogens and parasites in wild solitary bees and other pollinators (e.g. hoverflies) is even more limited. Recent work in Georgia, Germany, and Kyrgyzstan has shown that so-called honey bee viruses can infect and replicate in solitary bees (*Andrena haemorrhoa*, *Anthophora plumipes*, and *Osmia bicornis*) that are also native in England, and that transmission of these viruses between honey bees and wild solitary bees is likely (Radzeviciute *et al.* 2017). Similar studies in England identified honey bee viruses in hover flies, but did not demonstrate replication of the virus, which is needed to demonstrate actual infection (Bailes *et al.* 2018).

## 3.5.4 Knowledge gaps

- Determine the epidemiology of viruses and other pathogens in managed and wild pollinators, including directionality of transmission.

- Understand the impacts of emerging diseases in wild pollinator populations.

- Quantify the relative impact of pests and diseases on managed and wild pollinator populations, in comparison to, and in combination with, other drivers of population change.

- The current levels of pathogens and parasites in managed pollinators – honey bees, bumble bees, solitary bees, hover flies – and their potential risk to act as sources for disease emergence in managed and wild pollinators.

- Determine the effectiveness of current control and management strategies to prevent novel invasive pests establishing in England, in particular the Asian hornet, Tropilaelaps mites, and the small hive beetle, and devise effective controls where lacking.

# 4. Effectiveness of management and policy for pollinators

Authors: Lynn V Dicks, Deepa Senapathi, Katherine C R Baldock, Mark J F Brown

This section deals with main developments in the evidence base since Vanbergen *et al.* (2014) and includes: agriculture, nature conservation, urban and transport infrastructure, and beekeeping and pollinator management

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## 4.1 Rural: Agriculture – Lynn Dicks; and Nature Conservation – Deepa Senapathi

Vanbergen *et al.* (2014) did not explicitly cover the effectiveness of agricultural management and policy for pollinators, so here we summarise the most important sources of evidence from the last 20 years on the effectiveness of agricultural management and policy for mitigating and reversing pollinator declines.

### 4.1.1 Relevant agri-environment scheme and Ecological Focus Areas options

There is good evidence from around the world to show that wild pollinators use sown nectar flower mixtures, bare ground (as nesting sites), restored species-rich grassland and heathland, set-aside areas, nest boxes for cavity-nesting solitary bees (not for bumble bees), and uncropped naturally regenerated field margins. This evidence is summarised on the Conservation Evidence website ([www.conservationevidence.com](www.conservationevidence.com)) and has also been reviewed by Garibaldi *et al.* (2014). For the UK, 17 specific agri-environmental management options either known to, or widely considered to support pollinators were identified by Dicks *et al.* (2015), with an overview of the evidence supporting their benefits to pollinators and the resources they provide (see table 3, Dicks *et al.* 2015). All of these options are available under the English Countryside Stewardship Scheme (at mid tier, under the Wild Pollinator and Farm Wildlife Package, or Higher tier). Some of the options (fallow land, field margins, buffer strips and forest edges) are also included as Ecological Focus Areas (EFAs), obligatory on farms with >15 ha of arable land under the new Common Agricultural Policy. However, management to ensure benefit to pollinators is not so clearly prescribed in the EFAs policy. Of those EFAs known to have benefits for pollinators in terms of resource provision, only fallow land has been widely adopted by European farmers so far (Pe'er *et al.* 2016).

Organic farms generally have higher pollinator densities and numbers of wild pollinator species than non-organic farms (see figure 4.1), although this is not always true. Studies have found more flower resources within fields on organic farms, than on non-organic farms (Gabriel and Tscharntke 2007; Hardmann *et al.* 2016). This difference did not scale to farm level in the UK study, comparing organic farms to non-organic farms with agri-environment schemes, due to high flower densities in non-crop habitats on farms with Entry Level Stewardship and Conservation Grade schemes.

### 4.1.2 How effective has the creation of wild flower habitat in the UK been in improving the status of pollinators?

The UK Countryside Survey undertaken in 1990, 2000 and 2007 shows that plant species richness on arable and horticulture land increased by more than 30% between 2000 and 2007, partly due to an increase in sown wildflower field margins which are used by wild bees (Norton *et al.* 2012). This change has been associated with a slow-down in the loss of bee species-richness at a range of spatial scales after 1990 (1990-2009), compared to rates of loss in the previous eighty years (1920-1989) (Carvalheiro *et al.* 2013), as described by Vanbergen *et al.* (2014).

Figure 4.1 shows the positive effects of flower strips, naturally regenerated uncropped margins, extensively managed grassland and organic farming on the numbers of bees, hoverflies and butterflies found in those habitats, based on a meta-analysis of 71 European studies (Scheper *et al.* 2013). Each study compared

numbers of individuals or species, relative to a conventionally managed control site. These results represent insect densities, or numbers of insects using the created habitats as a resource. They cannot be directly interpreted as long-term benefits to pollinator populations, or reversal of pollinator declines. For this, measures of trends in numbers over time (over at least four years) or measures of reproductive success or survival are required. Empirical evidence (Blaauw and Isaacs 2014) and bumble bee population modelling (Iles, Williams, and Crone 2018) both indicate that four years of monitoring is a minimum time period for wild bee populations to respond to changes in agricultural land management.

Table: 4.1 - Management options that supply resources for pollinators. Options listed were identified as important or effective for pollinators by at least one of the following sources: Campaign for the Farmed Environment (Campaign for the Farmed Environment 2013); Entry Level Stewardship Handbook (Natural England 2013); evidence-based prioritisation of actions for bee conservation (Sutherland *et al.* 2012); habitat type identified with high floral resource or nesting suitability by Kennedy *et al.* (2013); habitat type identified with high mean numbers of nectar plants per plot by Smart *et al.* (2010). Adapted from Dicks *et al.* (2015)

| Management Option on Agricultural Land | Pollinator resources supplied |
|---|---|
| Nectar flower mixture | Food |
| Hedgerow management for landscape and wildlife | Food, nesting, refugia |
| Combined hedge and ditch management | Food, nesting, refugia |
| Management of woodland edges | Food, nesting, refugia |
| Supplement to add wildflowers to field corners and buffer strips on cultivated land | Food |
| Management of field corners | Food, nesting, refugia |
| Permanent grassland with very low inputs | Food, nesting, refugia |
| Ryegrass seed-set as winter/spring food for birds | |
| Legume- and herb-rich swards | Food |
| Unsprayed and/or unfertilised cereal headlands | Food, refugia |
| Selective use of spring herbicides | Food, refugia |
| Create patches of bare ground for ground-nesting bees | Nesting |
| Restore species-rich grassland vegetation | Food, nesting, refugia |
| Restore lowland heathland | Food, nesting, refugia |
| Provide set-aside areas in farmland | Food, nesting, refugia |
| Provide artificial nest sites for solitary bees | Nesting |
| Leave arable field margins uncropped with natural regeneration | Food, nesting, refugia |
| Increase the diversity of nectar and pollen plants in the landscape | Food |
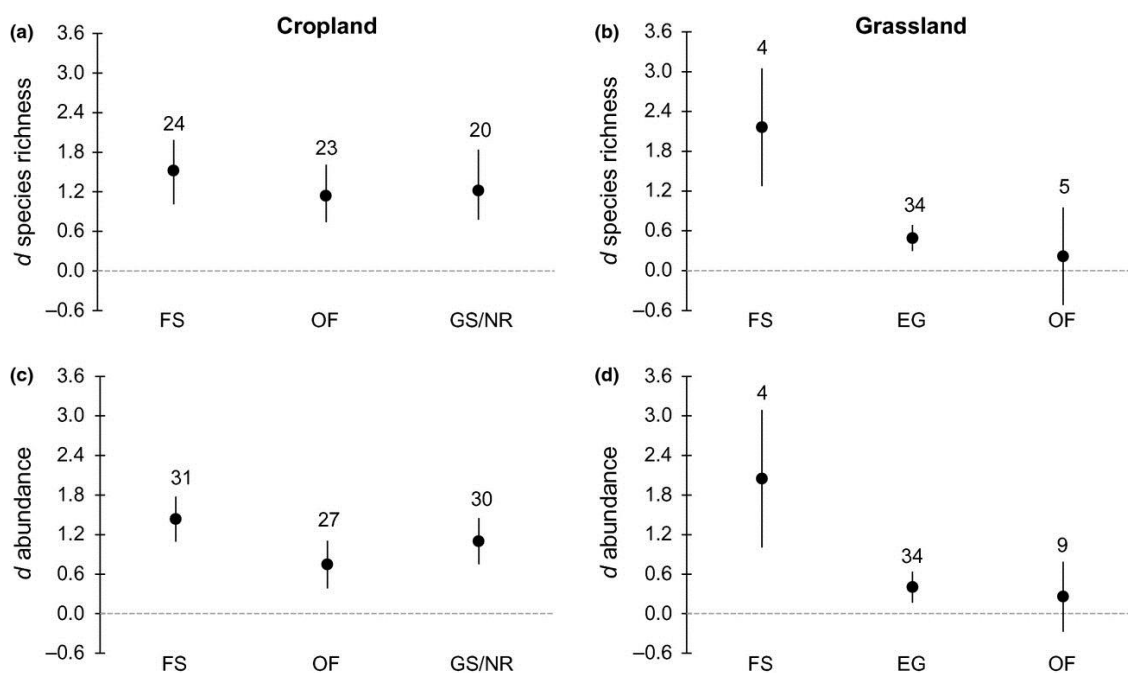
**Figure 4.1:** Effects of different types of agri-environmental measures on species richness (top) and abundance (bottom) of pollinators in croplands (left) and grasslands (right). Indicated are mean effect sizes (Hedges'd) with 95% bias-corrected bootstrap confidence interval (a) or parametric 95% confidence interval (b, c, d). Hedges'd is a standardised metric used to compare effects across studies with different outcome measures. It is related to the number of standard deviations away from the mean, and cannot easily be translated into actual numbers of species or insects. Numbers indicate the number of independent studies. FS: sown flower strip; OF: organic farming; GS/NR: grass-sown or naturally regenerated field margin or set-aside; EG: extensive grassland. Source: Scheper *et al.* 2013.

In recent years, evidence has emerged that enhancing flower-rich habitats in agricultural landscapes can lead to population-level benefits for wild bumble bees, in terms of enhanced reproductive success, bumble bee colony survival or population density. Carvell *et al.* (2017) show that for three common bumble bee species (*Bombus terrestris, B. lapidarius* and *B. pascuorum*) the survival of family lineages from the summer worker to the spring queen stage in the following year increases significantly with the proportion of high-value foraging habitat, including spring floral resources, within 250–1,000m of the natal colony. In a previous study, Carvell *et al.* (2015) showed that numbers of bumble bee reproductives (males or males and queens) tend to increase as flowers are added to a landscape. In a study of 19 farms in Hampshire and West Sussex (Southern England), Wood *et al.* (2015) showed that farms with Higher Level Stewardship agreements had higher floral resources, and this was associated with higher bumble bee nest densities for four common bumble bee species, compared to farms with Entry Level Stewardship agreements (Wood *et al.* 2015). Evidence for impacts of floral enhancement on populations of pollinator groups other than bumble bees is scarce.

Baude *et al.* (2016) estimated national nectar provision across the UK, using nectar productivity for 260 flowering plant species, and plant community data for different habitat types from the Countryside Survey. Habitat types included five agri-environment scheme options. The analysis showed that calcareous grassland, broadleaved woodland, neutral grassland and shrub heathland are the most important habitats for nectar provision. Four plant species – white clover *Trifolium repens*, Ling *Calluna vulgaris*, Marsh thistle *Cirsium palustre* and Bell heather *Erica cinerea* together produce over 50% of the UK's nectar supply, across all habitats. The overall contribution of agri-environment schemes to national scale nectar provision was low in 2007, when all relevant data were available.

Of the five types of agri-environment scheme options, nectar flower mixtures have the highest nectar productivity value (kg of sugars/ha/year), followed by enhanced grass margins. Nectar productivities of wild bird seed mixture, hay-making supplement and species-rich semi-natural grassland were much lower (Baude *et al.* 2016). Nectar flower mixture options are similar to hedgerows in terms of annual nectar productivity per

unit area, but in 2007 they covered a much smaller area (1,610 ha), and consequently contributed far less to the national nectar resources (0.1% of nectar supply comes from nectar flower mixtures compared to 3% from the 40,200 ha[5] of hedgerows in England (2016)).

The contribution of nectar flower mixtures and hedgerows to floral resource provision may have increased since 2015 due to the Countryside Stewardship Scheme's Wild Pollinator and Farm Wildlife Package. This has supported 13,000 ha of 'priority field margin options', which include flower-rich margins and pollen and nectar mixes[6]. Even if all the hectares of nectar pollen mix from 2007 had been lost, this could represent an eight-fold increase in flower-rich field margin habitat added under agri-environment schemes, but it would still only represent less than 1% of the national nectar provision for England according to Baude *et al*'s analysis (Baude *et al.* 2016).

National-scale calculations of floral resource provision have not yet been completed for pollen, which is the main protein source for bees and most hoverflies, and arguably more important to their reproductive success than nectar.

Dicks *et al.* (2015) combined estimates of pollen demand by wild bees with pollen supply from hedgerow and flower strip agri-environment options to calculate that 2% flower-rich habitat and 1 km flowering hedgerow per 100 ha of farmland, are sufficient to supply six common pollinator species with enough pollen to feed their larvae at lowest estimates, using minimum demand and maximum supply values for estimated parameters where a range was available. There was a very wide range of uncertainty. At the other end of the scale, high end estimates of pollen demand and low estimates of supply, suggested the six bee species would need 44% of the farmed landscape sown as well-managed nectar flower mix and 13.8 km of flowering hedge per 100 ha to meet their pollen demands through the season.

Carvell *et al.* (2017) showed that survivorship of bumble bee family lineages increased almost linearly with increasing proportions of flower-rich semi-natural habitat (including sown field margins) within 1 km of colony locations, in the range from 2-10% semi-natural habitat. With 2% flower-rich habitat, survival probability from summer worker to spring queen stage was less than 20%, rising to almost 60% when the landscape had 10% semi-natural habitat (figure 4.2).
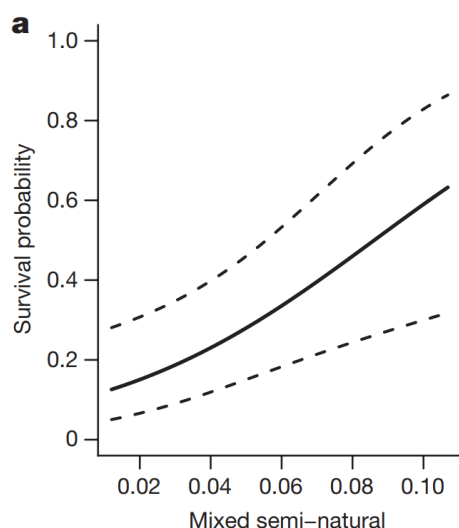


**Figure 4.2**: Effect of mixed semi-natural vegetation cover (including sown field margins) within 1,000 m of colony locations, on bumble bee family lineage survival from the summer worker to spring queen stage. Solid line shows model-fitted logistic regression. Dashed lines show 95% confidence intervals. Source: Carvell *et al.* 2017, Figure 2a.

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[5] Data from 2007 Countryside Survey
[6] Data from Natural England, submitted to Defra Pollinator Advisory Steering Group, 12 April 2018. Assumes the 13,000 ha does not include managed hedgerows.

## 4.1.3 What are the factors determining whether habitat creation is successful in supporting pollinators?

The factors determining the success of habitat creation at supporting pollinators are: the **magnitude, type and quality of the floral resources provided**, and **the landscape context** – particularly the complexity of the landscape in terms of habitat provision and the magnitude of increase in floral resources compared to the surrounding farmland (Scheper *et al.* 2013). The **potential exposure of pollinators to pesticides** through flowers planted in field margins alongside crops must also be considered.

### *4.1.3.1 Magnitude and quality of floral resources*

The density of floral resources in a habitat is almost always a key predictor of both abundance and species richness of the pollinator community found foraging there (Carvell *et al.* 2004; Potts *et al.* 2003; Roulston and Goodell 2011; Cole *et al.* 2017). For example, Garratt *et al.* (2017) found more than twice as many bumble bees near good quality hedges (defined by structure and woody species content) compared to poor quality ones in southern England. As discussed in section 4.1.2, it should be emphasized that such results represent behavioural responses of pollinators seeking out food resources, and do not necessarily reflect population increases. Provision of appropriate resources is key, and the types of flowers planted affect the type of pollinators that can benefit. In Wood *et al*'s Higher Level Stewardship farms in southern England, floral resources were higher than on Entry Level Stewardship farms (Wood, Holland, and Goulson 2015), but overall bee diversity was not enhanced (Wood, Holland, and Goulson 2015). The majority of solitary bee species present on these farms collected most of their pollen from plants that persist unaided in the wider environment, such as scentless mayweed *Tripleurospermum inodurum*, hogweed *Heracleum sphondylium* and dandelion *Taraxacum agg.* and not from those included in agri-environment schemes focused on pollinators (Wood, Holland, and Goulson 2017).

Some plant species are particularly important nutritionally for wild pollinators, either due to their time of flowering or their nutritional content. Woody plants, including trees, are particularly important sources of nectar and pollen for bumble bees in spring and early summer.

Carvell *et al.* (2017)'s data demonstrate a strong association between bumble bee queen survival and spring flower cover, largely provided by trees such as goat willow (*Salix caprea*) and woody hedgerow plants such as hawthorn *Crataegus monogyna* and blackthorn *Prunus spinosa*. Kamper *et al.* (2016) found that woody plants comprised 80% by volume of the pollen stores collected by 30 *Bombus terrestris* colonies in a mixed grassland and arable landscape in southern Germany in May. Key woody plant sources for bumble bee-collected pollen were *Acer* (sycamore/maple; 35% of pollen collected), *Lonicera* (honeysuckle; 15%) and Rosaceae (includes hawthorn and blackthorn; 13-15%). Very similar results, with the addition of willow *Salix* sp., have been found for honey bees in Wales (de Vere *et al.* 2017) and western France (Requier *et al.* 2015). The latter was a season long study that also revealed a key importance of agricultural weeds, particularly poppy *Papaver rhoeas*, to honey bee pollen supplies in early to mid-summer.

At the end of the summer, ivy *Hedera helix* is a key resource (Garbuzov and Ratnieks 2014) for many pollinators, including representing 89% of pollen collected by six honey bee hives in southern England in autumn.

Important elements of nutrition in flower resources are the overall protein content, the protein:lipid ratio of pollen and the specific amino acid composition of both pollen and nectar. These vary among plant species (Donkersley *et al.* 2017). While the specific amino acid and protein requirements for different pollinator species remain to be determined, there appears to be consensus that a diversity of floral species is needed to provide adequate nutrition to bees at different life stages (Roger *et al.* 2017; Donkersley *et al.* 2017; Alaux *et al.* 2010; Alaux *et al.* 2017). There is experimental evidence from honey bees that amino acids missing from diet are actively preferred by foragers (Hendriksma and Shafir 2016), although this has yet to be shown for any UK species of wild bee.

Some plants even confer particular health benefits for bees. There is emerging evidence, for example, that certain secondary metabolites in nectar (Richardson *et al.* 2015; Stevenson, Nicolson, and Wright 2017) or types of pollen diet (Di Pasquale *et al.* 2013) may reduce parasite loads or impacts of parasites in bees.

Management actions other than planting regimes, such as cutting or mowing dates, also influence floral resources substantially. For example, Staley *et al.* (2012) found that reducing the cutting frequency of hawthorn (*Crataegus monogyna*) hedges from every year to every 3 years resulted in 2.1 times more flowers. Pywell *et al.* (2011) demonstrated that summer cutting (in late May or June) and removal of cut material can both enhance overall floral resources, by extending the flowering season of nectar and pollen margins. These authors recommended that half the flowering area is cut in early summer. Advice on managing habitats for pollinators has been compiled in a freely available guidebook by the Centre for Ecology and Hydrology (Nowakowski and Pywell 2016).

Farmer experience and training also influence the quality of flower-rich habitats produced by agri-environment scheme options, with more experienced farmers producing habitats with greater numbers of flowers for pollinators (McCracken *et al.* 2015).

### 4.1.3.2 Landscape context

One general pattern is that landscapes of intermediate complexity, defined as 1-20% semi-natural habitat (referred to as 'simple' landscapes; Scheper *et al.* 2013), provide the greatest relative benefits in terms of use of these habitats by pollinators (figure 4.3). This pattern was predicted in theory (Tscharntke *et al.* 2005), and demonstrated by meta-analysis of empirical data from across Europe by Scheper *et al.* (2013) (figure 4.3). Agri-environment schemes for pollinators are likely to be ineffective in very intensively farmed 'cleared' landscapes (<1% semi-natural habitat). This is thought to be because there are few pollinating insects living in such landscapes to exploit the added resources, or because the landscape does not provide other essential resources such as nesting and over wintering sites.



**Figure 4.3:** The effects of agri-environmental measures on pollinator species richness and abundance, depending on landscape context. Structurally cleared landscapes: < 1% semi-natural habitat; simple landscapes: 1–20% semi-natural habitat; complex landscapes: > 20% semi-natural habitat. Indicated are mean effect sizes (Hedges'd) 95% CI. A mean effect size is considered significant when its CI does not include zero. Numbers indicate sample sizes.

Scheper *et al.* (2013) also defined 'ecological contrast' as a key factor influencing the effectiveness of agri-environment scheme habitat additions for pollinators. In this context, ecological contrast is the increase in floral resources generated by a newly created habitat, compared to the level of floral resources in the

surrounding farmland. The impact of agri-environmental measures on bee species richness and abundance increased significantly with increasing difference in plant (forb) species richness between the agri-environment treatment and control crop fields (Scheper *et al.* 2013). Similarly, a study of hedgerows in Southern England (Garratt *et al.* 2017) found greater numbers of wild bees (but not hoverflies) on or near hedgerows with lower local proportions of semi-natural habitat (across a range from 0-40% semi-natural habitat within a 500m radius), indicating hedgerows may be a more valuable forage resource for bees in more intensive landscapes lacking semi-natural habitat.

### 4.1.3.3 Pesticide exposure

It is possible that flowering resources placed alongside crop fields increase exposure of pollinators to pesticides. Evidence has recently emerged that flowers in field margins contain levels of systemic neonicotinoid insecticides comparable to or even higher than those in the treated crop themselves (Mogren and Lundgren 2016; Botías *et al.* 2015; Botías *et al.* 2016). Other pesticides, including fungicides, are also found in field margin flowers (David *et al.* 2016) and in-field weeds used as forage sources by bees (Long and Krupke 2016). The actual impact of these routes of exposure on pollinator populations is unclear (see Drivers section 3.2).

## 4.1.4 How successful are innovations to boost nesting opportunities for wild pollinators?

There is good evidence to show that wild ground-nesting solitary bees use freshly scraped bare ground for nesting in heathland and grassland contexts, and that cavity-nesting solitary bees will use solitary bee nest boxes, sometimes known as bee hotels, in many different contexts, including agricultural land (Dicks, Showler, and Sutherland 2010).

Nesting habitats for bumble bees include rough grassland for surface-nesting species (e.g. *Bombus pascuorum*), rodent holes for ground-nesting bumble bees (e.g. *Bombus terrestris*) and above-ground cavities, in trees for example, for aerial nesters (e.g. *Bombus pratorum*).

Studies of adding bare ground as a nesting resource for bees are limited to a small number of trials in heath and grassland, none in the UK (Dicks, Showler, and Sutherland 2010). These consistently show that artificially exposed areas of bare soil can be successfully colonised by ground-nesting solitary bees and wasps in the first or second year. We know of no scientific evidence testing this intervention in an agricultural context. The Irish 'Green, Low-carbon, Agri-environment Scheme (GLAS)' includes an option to place a tonne of builder's sand in the corner of a field for ground-nesting bees. The effectiveness of this measure is unknown.

There does not seem to be evidence from Europe for the effect of creating areas of bare ground on bee populations or communities on a larger scale, or on local pollination services, although this is frequently suggested as an innovation to support pollinators and enhance pollination (Dicks *et al.* 2015). In the USA, the ground-nesting alkali bee *Nomia melanderi* has been sustainably managed in large and growing nesting aggregations for decades, for its pollination services (Cane 2008). A current research project at the University of Reading is exploring the effectiveness of providing bare ground for solitary bees in UK apple orchards (BBSRC grant code: BB/P003664/1).

We also know of no specific research on the distribution or enhancement of nesting resources for ground-nesting in UK agricultural landscapes. In California, recent studies have found that ground-nesting bees nest throughout arable fields (in this case sunflower) but prefer to nest in field boundaries (Sardinas *et al.* 2016). Boundaries without hedgerows harboured more ground nesting bees than those with hedgerows (Sardinas, Ponisio, and Kremen 2016). Such research in UK environments will be key to designing appropriate conservation actions to support ground-nesting bees.

Nest box provision (e.g. hollow reed stems, cardboard tubes, drilled wooden boards) can increase the numbers of cavity-nesting solitary bees over time. Several studies have shown that the number of occupied solitary bee nests can double with repeated nest box provision for cavity-nesting bees at a given site (Dicks, Showler, and

Sutherland 2010; Dainese *et al.* 2018), while Moron *et al.* (2014) showed population growth rates of 2 to 5 in red mason bees *Osmia bicornis* using nest boxes in unpolluted sites in the UK and Poland, where 2 is a doubling of number of females in a cavity nest box after one year.

Bumble bee nest boxes are not effective in the UK, as the uptake rates are extremely low (0-3%) (Lye *et al.* 2011; Dicks, Showler, and Sutherland 2010). It is not clear whether the lack of effectiveness represents poor or inappropriate design or placement of bumble bee nest boxes, which could be improved upon, or whether natural nest sites are not limiting, so bumble bees do no need to use nest boxes.

## 4.1.5 Which aspects of recent land management in the UK are more or less important for pollinator status?

### 4.1.5.1 England

Based on the analysis of total nectar resources from different habitat types above, the addition of flower-rich field margins (such as nectar-pollen strips) through agri-environment schemes, whilst clearly effective at enhancing local pollinator numbers in landscapes of intermediate complexity, **is not at sufficient scale, or appropriately targeted, to reverse observed declines in wild pollinators**.

The pollinator species known to be declining tend to be habitat specialists, or those with specialised diets (Carvalheiro *et al.* 2013). For example, Dicks *et al.* (2015) identified a set of English wild bee species associated with farmland and known to be declining. Half (six) of the selected declining bee species are specialised in their choice of pollen sources (choosing yellow Asteraceae, Fabaceae or a few species in the Dipsacaceae). Sown nectar flower strips in field margins will not support these declining species, unless they supply the specific flowering resources they need, specified in Table S1 of Dicks *et al.* (2015).

There remains, however, a strong argument for continuing to support sown flower strips as an intervention to enhance local abundance of common pollinators and support crop pollination services (Kleijn *et al.* 2015).

The available evidence implies that **large-scale protection and restoration of flower-rich semi-natural habitats** such as grasslands, heathlands and broadleaved woodland are likely to be more important for improving the status of pollinators. Considering what is known about the influence of landscape structure on the effectiveness of pollinator conservation actions (Scheper *et al.* 2013), and the effects of the proportion of flower-rich habitat at landscape scale on bumble bee survival between years (Carvell *et al.* 2017), efforts to protect and restore these habitats would likely be most effective if targeted to landscapes with between 1 and 10% of pre-existing semi-natural habitat. The spatial extent of crop and wild flower pollination benefits from larger patches of semi-natural habitat are likely to be limited to a few kilometres or less around each patch (Ricketts *et al.* 2008; Garibaldi *et al.* 2011; Kennedy *et al.* 2013).

In addition, **large-scale changes to management of existing hedgerow, field margin, improved grassland and road verges**, to reduce cutting, fertilizer and herbicide use, would allow existing flowering species that support a wide range of wild pollinators to thrive, such as the plant species identified by Wood *et al.* (2016) for solitary bees. These habitats are known to be extremely important for pollinators (Cole *et al.* 2017; Baude *et al.* 2016; Hanley and Wilkins 2015). For example, Hanley and Wilkins (2015) found twice as many bumble bees foraging on the road side of hedgerows than on the arable field side of hedgerows, in a study in southwest England. In low intensity farming systems of the Somerset Levels, Osgathorpe *et al.* (2012) found track edges and road verges were particularly important sources of forage for long-tongued bumble bees. Cattle-grazed pasture in Somerset and winter-grazed pasture in the Outer Hebrides were also notable sources of bumble bee forage.

### 4.1.5.2 Scotland, Wales, Northern Ireland

As for England, it is likely that sown flower-rich field margins are at insufficient scale and not correctly targeted to influence the status of pollinators at national scale. However, we do not have analyses of the supply of nectar resources from agri-environment options from these countries to support this argument.

A landscape-scale study of an entire catchment in Ayrshire, Scotland, Cole *et al.* (Cole *et al.* 2017) identified road verges and  buffer strips along waterways as key habitats for butterflies, bees and hoverflies, as a result of these habitats having the most abundant and diverse floral sources.  Open scrub habitats were also important for butterflies.

### 4.1.5.3 Approaches to reducing pesticide use

Pesticides (insecticides, fungicides and herbicides) pose a significant threat to both wild and managed pollinators (see section 3.2), therefore actions to reduce their use must be considered as actions to conserve pollinators. It is not clear how important these actions are, relative to actions focused on enhancing floral and nesting resources through habitat management and restoration.

A range of responses to reduce pesticide impacts on pollinators were presented in the IPBES report on pollinators and pollination (IPBES 2016a). They include risk assessment, regulation, labelling, training and education of pesticide users, and bans or moratoria. A lack of long-term pollinator monitoring so far means there are no reliable data to indicate the impacts of these actions on pollinators in the UK. However, there is some evidence that restrictions reduced the risk to pollinators between 2002 and 2008. Based on risk indicators, Cross and colleagues found a decrease in the average environmental risk of pesticides per hectare for fruit and arable crops between the first introduction of risk-based regulations in 2002, and 2008 (Cross 2013; Cross and Edwards-Jones 2011).

A recent study (Goulson *et al.* 2018) interprets data on UK pesticide usage from 1990-2015 in terms of the potential toxicity of applied pesticides to honey bees (number of honey bee LD50s applied to UK farmland in total). It shows a substantial increase in potential toxicity from 2007 onwards, due to the increased use of neonicotinoids. This is not a measure of risk, since it does not incorporate probability of exposure.

Integrated Pest Management (IPM) has been specifically highlighted as an action for pollinator conservation (Dicks *et al.* 2016). IPM combines pest monitoring with a range of pest control methods, such as crop rotation, field margin management, and biological control; pesticides are used as a last resort, only when other strategies are insufficient. IPM can decrease pesticide use and reduces risks to non-target organisms, so it should be linked to pollinator health and pollination.

IPM is promoted in UK policy through the National Action Plan for the Sustainable Use of Pesticides (mandated by the Plant Protection Products (Sustainable Use) Regulations 2012) and the National Pollinators Strategy for England. The National Farmers Union has developed a self-assessment tool for farmers – the Integrated Pest Management Plan – as a voluntary measure. This has been completed by 16,820 farmers and growers, covering approximately 4.4 million ha of farmland (25% of UK total agricultural area). These plans should provide data on the uptake rates of specific IPM practices, although the data are not currently available for scrutiny. There is also no specific research on the impact of implementing IPM at farm scale on local pollinator populations or communities.

### 4.1.6 Is there evidence that different management actions would now benefit the status of pollinators or pollination service in the UK?

Recent reviews of policy-relevant science have highlighted the potential for **ecological intensification** of agriculture to mitigate the drivers of pollinator decline in intensive agricultural landscapes (IPBES 2016b; Kovács-Hostyánszki *et al.* 2017). Ecological intensification, as defined by Bommarco *et al.* (2013) and Tittonell (2014), involves actively managing farmland to increase the intensity of the ecological processes that support production, such as biotic pest regulation, nutrient cycling and pollination. It means making smart use of nature's functions and services, at field and landscape scales, to enhance agricultural productivity, and reduce reliance on agrochemicals and the need for further land-use conversion. Ecological intensification is a knowledge-intensive process that actively engages farmers in promoting biodiversity beneficial to agricultural production, through management practices such as intercropping, crop rotations, farm-level diversification and reduced agrochemical use.

### 4.1.7 Nature Conservation

Pollinator species rarely rely solely on a single plant species for either food or nesting resources; neither are all flower visitors effective pollinators. Since the flowering period of a single plant species is often short in comparison to the activity period of pollinators (Blüthgen and Klein 2011), pollinators depend on a range of plants which are more readily provided in (semi-) natural habitats. Actions aiming to conserve pollinators should therefore consider the wider diversity and landscapes which sustains these species:

Nature conservation for pollinators can focus on supporting single species or single habitats types as well as wider conservations across landscapes. Actions aimed at protecting priority habitats such as heathlands which may be species poor, may however support more specialized plant–pollinator interactions (Ballantyne, Baldock, and Willmer 2015), and help protect specific pollinator species.

A recent global meta-analysis found that habitat restoration in non-agricultural settings such as heathland, scrub or woodland had overall positive effects on wild bee abundance and species richness, even though the restoration activities were not focused on pollinators specifically (Tonietto and Larkin 2017). Restoration techniques were usually focused on restoring the plant community, and included burning, grazing, seeding and removing invasive plants.

#### 4.1.7.1 Reintroduction of Bombus subterraneus

Declines in populations of bumble bees have recently become a cause for global conservation concern, both because of their intrinsic biodiversity value and the ecosystem services that they provide (Vanbergen *et al.* 2013). The Short-haired bumble bee (*Bombus subterraneus*), a widespread palearctic species (Nieto *et al.* 2014), native to the UK, was once widespread across England. Prior to the 1970s, this species was abundant in Deal and Dover in Kent, in Suffolk and in other localities in Southern and Eastern England (Sladen 1912, cited in Benton 2006). Last recorded near Dungeness in 1988, this species was declared extinct in 2000 (Gammans and Allen 2014).

In 2007, Natural England proposed the reintroduction of the short-haired bumble bee to the UK (Vaughan-Higgins *et al.* 2016) and in 2009 the Short-haired bumble bee partnership was formed between Natural England, the Bumble bee Conservation Trust, Hymettus and the RSPB. Funded by multiple partners, this project's initial attempts to reintroduce this species from New Zealand populations failed due to seasonal differences as well as high inbreeding within the source population. However, the County of Skåne in southern Sweden was identified as an alternate viable source of the bees and in Spring 2012, up to 100 queen bees were collected from two linear-transects in Skåne. Disease risk analysis was conducted and disease risk management plans constructed to design a capture-quarantine-release system that minimised the impacts on both the

bumble bees and on their natural parasites (Brown *et al.* 2017). Once the requisite quarantine period and health checks were passed, five releases took place between 2012 and 2016 within the release areas of Dungeness and Romnesy Marsh.

To date the project has created, advised and assisted in the management of over 1,300 hectares of flower rich habitat within the release area of Dungeness and Romney Marsh (BBCT 2018). The created habitat continues to be assessed through annual wild flower surveys and bumble bee transects. Recordings within the Dungeness and Romney Marsh area include the Ruderal bumble bee (*Bombus ruderatus*) – recorded on the reserve after a ten year absence; Brown-banded carder bee (*Bombus humilis*) and Moss carder bee (*Bombus muscorum*) and the Red-shanked carder bee (*Bombus ruderarius*) in addition to the target species (BBCT 2018).

Since 2017 the project has focussed on establishing whether a Short-haired bumble bee population has established by completing genetic analysis of all Short-haired bumble bees found. The results are to be reviewed in 2020 and decisions will be made then as to whether further reintroductions are required (BBCT 2018).

### 4.1.7.2 Other conservation initiatives

In terms of wider nature and nature conservation, projects like the Coronation Meadows and Save our Magnificent Meadows are crucial in conserving flower rich habitats and contribute to the National Pollinator Strategy by restoring and conserving some of our best species-rich meadows and creating new meadows. B-Lines is another nationwide initiative to restore and create a series of wildflower-rich habitat stepping stones aiming to link existing wildlife areas together to create a pollinator friendly network of habitats that will weave across the British landscape.

The Glastir Small Grants is a programme of capital works available to farming businesses across Wales to carry out projects that will help to improve and maintain the traditional landscape features in Wales, and provide habitat linkage for pollinating insects. Defra grants to Local Nature Partnerships projects in Hertfordshire, Lancashire, Lincolnshire, Durham and Surrey support schools and museums to develop pollinator friendly courses, promote ecological understanding and awareness as well as create roadside habitats for our bees and butterflies

## 4.1.8 Knowledge gaps

- Analysis and mapping the contribution of different habitat types and particular plant species to the provision of pollen resources, and to the specific nutritional requirements of wild pollinators, at landscape scale.
- UK based research on the effectiveness of adding nesting resources for bees, especially nesting habitats, to support their populations and enhance the pollination services they provide. This includes cavity-nesting and ground-nesting bees, which require different resources.
- Monitoring population-levels impacts of the actions for pollinators listed in table 4.1 – including measurement of pollinator numbers **that reflect survival and reproductive success**, rather than the numbers of foraging individuals at a particular place; and conducted over **at least four years**, to allow time for populations to respond.
- Up-to-date spatial analysis of UK farmed landscapes, to quantify the proportion of flower-rich semi-natural habitats and identify areas with between 1 and 10% of such habitats, where pollinator conservation actions can be targeted to increase the probability of effectiveness.
- Quantification of the effects of Integrated Pest Management (IPM) and other changes to the pesticide regime, on pollinator diversity, abundance and health in farmland.
- Published evidence of initiatives in areas excluding agricultural land and urban environments.
- There is a lack of evidence on effectiveness of conservation initiatives.

## 4.2 Urban and transport infrastructure – Katherine Baldock

Urban areas are comprised of multiple land uses and habitat types.  Broadly, urban greenspace falls into two main categories: (1) publicly accessible greenspace managed by local authorities (public parks, amenity grassland including playing fields, road verges) and (2) privately owned greenspace (residential or other private gardens).  Other land uses or habitats that could be used by pollinators include allotments, cemeteries and churchyards, school playing fields, planters in built up areas, green roofs, urban nature reserves, industrial estates, brownfield sites, and university and college campuses.  Transport infrastructure comprises road and railway verges, and green space at airports.  Below we summarise the evidence for the effectiveness of different management approaches for pollinators.  As Vanbergen *et al.* (2014) did not explicitly cover the effectiveness of management and policy for pollinators in urban areas and transport infrastructure here we consider all available published studies, including those published prior to 2014.

### 4.2.1 Management of urban areas for pollinators

Different urban land uses vary in their availability of forage resources for pollinators, and numbers of pollinator visits to flowers. In a study of 360 sampling sites across four UK cities, Baldock *et al.* (2019) found that allotments and residential gardens contained higher floral abundances per unit area in comparison to the other seven land uses studied (cemeteries and churchyards, urban nature reserves, parks, road verges, other greenspace areas, pavements, manmade surfaces).  This pattern was driven by the higher floral abundance of non-native plant taxa in allotments and gardens; native floral abundance did not differ significantly among most land uses.

The study by Baldock *et al.* (2019) suggests that allotments are important land uses for pollinators in cities. Numbers of bee visits were higher in allotments than in all other land uses except gardens, and numbers of hoverfly visits were higher in allotments compared to road verges, pavements and other greenspace areas. A modelling approach incorporating plant-pollinator interaction data was used to test the effect of increasing the area of each land use in turn to four cities, measuring how the additional area of each land use affected plant-pollinator community robustness at a city scale. The model predicted that increasing the area of allotments resulted in the greatest increase in the plant-pollinator community robustness.

The results also suggest that residential gardens are an important urban land use for pollinators.  Numbers of bee visits were higher in gardens compared to all land uses except allotments, and gardens had more hoverfly visits in comparison to road verges, pavements and other greenspaces.  Gardens comprised large areas (24-36% of the area of the study cities), thus when the data were scaled up to city level, gardens hosted an estimated 54-83% of pollinator flower visits in cities.  The study also considered the effect of household income, finding that numbers of pollinator visits were higher in gardens located in neighbourhoods with higher household incomes.  Similarly, the numbers of flowering plant taxa and the numbers of flowers were higher in gardens, which suggests that socio-economic factors can influence the quality of floral resources available for pollinators in urban gardens.

The authors highlight management of public greenspace as an important pollinator conservation opportunity in urban areas.  Parks, road verges and other greenspace collectively comprised 27-35% of the study cities however these land uses were estimated to host far fewer pollinator visits at a city scale compared to gardens, which made up a similar area in cities. Modelling the effect of increasing the floral abundance of three common native plant species (*Taraxacum* agg., *Bellis perennis* and *Trifolium repens*) in parks, road verges and other greenspaces, showed that this led to increased plant-pollinator community robustness at a city scale. As the abundance of all three plant species is likely to be increased by reducing mowing, this suggests that reducing mowing frequencies is likely to benefit pollinator communities.

## 4.2.2 Mowing of public greenspace, road verges and residential gardens

A review by O'Sullivan *et al.* (2017) summarises the evidence for the effect of urban road verge management on biodiversity and ecosystem services, including pollinators. However, few studies have examined the effect of different mowing regimes on pollinators directly.  In a comparison of five mowing treatments on rural grass verges in the Netherlands Noordijk *et al.* (2009) found that mowing twice a year (compared to no management or one mow a year) with hay removal was the best approach to maximise floral abundance, insect abundance and insect flower visits throughout the growing season.  The early summer cut was considered to be important in contributing to re-flowering of plants later in the growing season, however since these plots contained no flowers for a period after mowing Noordijk *et al.* (2009) recommend a rotational mowing scheme to maximise floral availability.  Since urban road verges in the UK are typically mown every 2-4 weeks (O'Sullivan *et al.* 2017) they are likely to provide fewer floral resources for pollinators than less frequently mown rural verges.

There are a limited number of studies that have considered the effect of different mowing regimes in an urban context. Garbuzov *et al.* (2015) compared mowing regimes in a single UK park, finding that reduced mowing was associated with increased floral abundance and numbers of insect flower visits.  Lerman *et al.* (2018) examined the effect of different lawn mowing frequencies (every 1, 2 or 3 weeks) on bee abundance in 16 suburban residential gardens in Massachusetts, USA, finding that lower mowing frequencies were associated with higher floral abundance and bee abundance, suggesting that mowing lawns less frequently is beneficial for pollinators. Although these studies contrast with the findings of Noordijk *et al.* (where increasing mowing frequency was the optimal approach) the cutting frequencies in these studies was considerably higher than in Noordijk *et al.*'s study of rural road verges.

Garbuzov *et al.* (2015) also examined the effect of ceasing regular mowing in the summer rather than continuing into the autumn, considering four treatments in total: regular mowing every 2 weeks all spring and summer, regular mowing until 2 June, regular mowing until 5 July, and no mowing.  Their findings suggest that stopping mowing in June leads to higher numbers of flowers and insect flower visits in comparison to mowing into the autumn, although stopping mowing in July is not beneficial for increasing flower and visit numbers. Not mowing at all resulted in the highest numbers of flowers and flower visits.  It should be noted, however, that this study sampled in only one urban park.

## 4.2.3 Management of railway embankments

Several recent studies from Poland have explored the pollinator communities associated with railway embankments in agricultural landscapes. Moron *et al.* (2014) found that the occurrence of bare ground had a positive effect on bee species richness and abundance, although a negative effect on butterfly populations. Butterfly species richness and abundance was positively affected by the species richness of native plants, while bee species richness was positively affected by the species richness of non-native flowering plants. Shrub density had a negative effect on bee species richness and abundance, thus the authors recommend controlling shrub encroachment as a conservation strategy. Bee and hoverfly species richness were positively related to wood cover in the surrounding landscape. Moron *et al.* (2017) found that railway embankments function as corridors for at least some pollinator taxa. Wrzesien *et al.* (2016) evaluated bee forage flora composition and diversity on railway embankments, finding that they represented valuable refuge areas for bee forage plants and pollinators. Their study included tracks associated with three different traffic volumes (high volume high speed railway, intermediate volume, low volume tourist track) and found that the highest forage plant diversity was associated with intermediate traffic volume. Possible reasons the authors suggest for this include application of herbicides along embankments with high traffic volume and more extensive mowing along the low volume tracks.  The study also found that forage plants differed between the top and bottom of embankments and suggest that mowing is an important management approach for promoting forage plants, especially on tracks with low and high traffic volume, to prevent the spread of aggressive and invasive species.

### 4.2.4 Addition of floral resources

The addition of flower-rich plantings (often referred to as 'meadows') to public greenspace areas has been used as a management tool to improve urban areas for pollinating insects.  These meadows can provide large quantities of additional floral resources in the form of pollen and nectar (Hicks *et al.* 2016). While Hicks *et al.* (2016) found that both perennial meadows composed of native species and annual meadows composed of mainly non-native species significantly increased the pollen and nectar content compared to amenity grassland, the perennial native meadows produced up to 20 times more nectar and up to 6 times more pollen than annual meadows of equivalent size.  They also found that weed species growing in the meadows contributed high quantities of pollen and nectar, with dandelions (*Taraxacum* agg.) being one of the most important pollen and nectar contributors among the species commonly considered as weeds.

Blackmore *et al.* (2014) compared plots sown with wild flowers to control plots of amenity grassland, finding that sown plots had 25 times more flowers, 50 times more bumble bee visits and 13 times more hoverfly visits. Hoverflies were more abundant in plots in their first flowering year, suggesting a preference for shallow annual flowers.

### 4.2.5 Choice of floral species for plantings

Floral species can vary in their attractiveness to pollinating insects, and different species can be attractive to different pollinator groups. The amount of pollen and nectar produced varies among plant species (Comba *et al.* 1999, Corbet *et al.* 2001, Baude *et al.* 2016, Hicks *et al.* 2016), therefore the choice of species to plant for pollinators should focus on those that produce larger quantities of pollen and nectar that are accessible to UK pollinators over an extended flowering period.  While recent studies have sampled nectar and pollen for UK native plant species and some non-native species (Baude *et al.* 2016, Hicks *et al.* 2016) little information exists on the quantity and quality of pollen and nectar for the majority of horticultural plants (this term is used here to refer to plants grown for any use other than agriculture).

Horticultural plants can be of native or non-native origin to the UK and choice of plant origin for plantings could affect how beneficial they are for pollinators. Salisbury *et al.* (2015) examined insect flower visitation to plants belonging to three treatment groups: UK native species, near-native (occurring naturally only in the Northern Hemisphere but not native or naturalized in the UK) and exotic (occurring naturally only in the Southern Hemisphere). While overall more pollinator visits were recorded on native and near-native compared to exotic plants there were differences among pollinator groups. More hoverflies were recorded on native plants, whilst more honey bees were recorded on near-natives. There were no differences in the abundance of long-tongued bumble bees or solitary bees among the treatments, whereas short-tongued bumble bees were more abundant on native and near-native plants. Numbers of pollinator visits corresponded to the peak flowering period of the treatment, thus the exotic plants had fewer visits in early summer but comparatively more, later in the season. Salisbury *et al.* advise that whilst plantings comprising natives and near natives might attract more pollinators, using exotic plants to extend the flowering season could be beneficial for pollinators later in the year.

Hanley *et al.* (2014) also explored visitation patterns to plants of different origin (British, Palaearctic, non-Palaearctic). Although they found inconclusive results depending on which plants were included in analyses, their results suggest that dietary specialist bumble bees (those with longer tongues, *Bombus hortorum* and *Bombus pascuroum*) prefer plants found in their own biogeographical range, whereas shorter tongued species showed little preference for British or Palaearctic plants. They also suggest that non-horticultural 'weed' species are important for urban pollinators.

Baldock *et al.* (2019) found that across the nine land uses sampled, both native and non-native plant taxa were found to be important pollinator forage plants, with four native species (*Cirsium arvense*, *Heracleum sphondylium*, *Ranunculus repens*, *Taraxacum* agg.) and one non-native species (*Borago officinalis*) receiving more pollinator visits than expected given their floral abundances.

Some horticulturally modified plant varieties may be less beneficial for pollinators than non-modified varieties. Comba *et al.* (1999) found that cultivars of six garden plants with spurless, doubled, peloric or enlarged flowers generally contained less nectar than cultivars with near-original flower types, often had less accessible nectar and had fewer flower visitors. Corbet *et al.* (2002) also found that double flowered varieties of several plant species produced little or no nectar and received few pollinator visits.

Although studies have examined pollinator visitation to garden plants (see also Garbuzov *et al.* 2014, 2017), the pollen and nectar content of, and pollinator visitation rates to flowers of many horticultural plants are unknown. Furthermore, while many lists of plants that are recommended for pollinators exist, e.g. the Royal Horticultural Society's Plants for Pollinators, none are based on published data (Garbuzov & Ratnieks 2014).

## 4.2.6 Use of pesticides

Several recent studies have examined the effect of pesticides in urban settings. Larson *et al.* (2013) found that neonicotinoid pesticide use in urban lawns negatively affected bumble bee colony growth and new queen production when applied to blooming plants. Muratet and Fontaine (2015) found a negative correlation between butterfly and bumble bee abundance and use of insecticides and herbicides in a study using citizen science data from French gardens. The study also showed that the negative effect of insecticides was stronger in highly urbanised areas. Botías *et al.* (2017) found that bumble bees in ornamental urban gardens had detectable levels of neonicotonoid insecticides (thiamethoxam, clothiandin, imidacloprid, thiacloprid, acetamiprid) and fungicides (carbendazim, boscalid, tebuconazole, prochloraz, metconazole, trifloxystrobin, fluoxastrobin, although concentrations and detection frequencies were higher in bees collected from farmland sites. See section 3.2 for an overview of pesticides as drivers of pollinating insects decline.

## 4.2.7 Green roofs

A green roof is a roof of a building that is partially or completely covered with vegetation and a growing medium. Several recent studies have examined pollinators on green roofs. Tonietto *et al.* (2011) found that native bees are present on green roofs in Chicago, although at lower abundance and diversity than in nearby reference habitats (tallgrass prairie natural areas and city park green spaces).

MacIvor (2016) monitored cavity-nesting bees and wasps in human-made trap nests (also known as 'bee hotels', see 4.2.8) on vegetated and non-vegetated roofs in Toronto, Canada, finding 27 species nesting on rooftops. Building height was negatively correlated with the abundance of brood cells provisioned in trap nests. Decreased species richness and abundance was related to a decline in green space area within a 600 m radius of the rooftop. The authors recommend using trap nests to support populations should focus on low- and mid-rise buildings (<5 building levels).

Ksiasek *et al.* (2012) examined pollinator visits to, and pollination success of, nine native Illinois prairie species on green roofs in Chicago. Although numbers and diversity of bees on green roofs were lower compared to the ground level, seed set was still high suggesting that the insects that are present provide sufficient pollinator services for many native plants.

## 4.2.8 Nesting sites

There are few studies that have considered bee nesting sites in urban areas.  UK gardens have been found to contain greater density and survival of bumble bee nests compared to agricultural and woodland habitats (Osborne *et al.* 2008, Goulson *et al.* 2010).

Artificial nesting sites (also known as trap nests or 'bee hotels') can be used to boost nesting opportunities for cavity-nesting bee and wasp species. Research on their effectiveness is limited; MacIvor & Packer (2015) surveyed almost 600 bee hotels set up for a period of 3 years in Toronto, Canada. The type of urban green space the bee hotel was located in was an important factor in the abundance of bees or wasps reared from the bee hotel. Native bee abundance was greatest for bee hotels in residential gardens, the abundance of introduced bee species was greatest on rooftops, and to a lesser extent community gardens, native wasp abundance was greatest in urban parks.

## 4.2.9 Knowledge gaps

- No studies have examined the effect of different mowing regimes on pollinators in urban road verges and none in rural settings in the UK.

- Studies of the pollinators and forage plants on railway embankments in the UK.

- An understanding of whether floral plantings in urban areas increase pollinator populations or increasing floral abundance acts as a "sink" drawing pollinators from other foraging sites.

- The effect of planting non-native taxa on pollinators and the ecosystem more widely.

- Information on the nectar and pollen provided by garden plants and their contribution to pollinator diets.

- Lists of plants recommended for pollinators based on peer-reviewed data rather than expert knowledge alone.

- Further research on the effect of pesticides and other chemicals on pollinators in urban settings.

- Further research on the benefits of green roofs and green walls for pollinators.

- A greater understanding of nesting sites used by pollinators in urban landscapes.

- There are a number of urban land uses for which we currently have little data on their suitability for pollinators or the effect of different management approaches: railway embankments, industrial areas, airports, university and college campuses.

- Data on beneficial management approaches of urban areas for pollinators is limited, with few studies conducted in the UK. Information on pollinators found in brown field sites, industrial areas and railway embankments is particularly lacking.

## 4.2 Beekeeping and pollinator management – Mark Brown

A range of pollinators – including honey bees, bumble bees, and solitary bees – are managed in England (and the rest of the UK) for honey production, wax production, and pollination services. The practices and policies which underpin these pollinators will impact bee health of both managed and wild pollinators. As the health of these managed pollinators is largely governed by policy and the practice that results, this section focuses on relevant legislation and its effects.

### 4.3.1 Honey bees

The import of honey bees into the EU, and trade within the EU, are governed by Council Directive 92/65/EEC and Commission Regulation (EU) No. 206/2010, both of which were specifically designed to control the spread of pests and diseases. In particular, they aim to prevent the introduction of the small hive beetle (*Aethina tumida*) and the Tropilaelaps mite (*Tropilaelaps* spp.) (see section 3.5) into the EU, and control the spread of American Foulbrood (*Paenibacillus larvae*) (see section 3.5). The arrival and apparent establishment of the small hive beetle in Italy (Mutinelli *et al.* 2014, Rivera-Gomis *et al.* 2017) demonstrates how the enforcement of such regulations can fail.

In England, the Directive and Regulation described above are implemented through the Trade in Animals and Related Products Regulations (TARPS) 2011 (Statutory Instrument No. 1197). Further legal regulation of bee pests and diseases is laid down in the Bee Diseases and Pests Control (England) Order 2006 (Statutory Instrument No. 342). Animal Plant and Health Agency (APHA) inspectors are authorised under the Bees Act 1980, the Bee Diseases and Pests Control Order 2016, and the TARP regulations 2011 (see above) to enter and inspect premises, destroy colonies infected with American Foulbrood and destroy or treat colonies infected with European Foulbrood (*Melisococcus plutonius*) (see section 3.5).

The National Bee Unit (APHA) works together with national (British Beekeeping Association) and local beekeeping associations to train beekeepers in effective honey bee management practices, to survey honey bees in England for health and disease, and to manage disease outbreaks. The incidence of the notifiable diseases American Foulbrood and European Foulbrood have remained largely stable over the last five years (2013-2017; http://www.nationalbeeunit.com/public/BeeDiseases/trendDiseaseChart.cfm), demonstrating the effectiveness of current policy and practice in this area. An ongoing campaign was launched in 2016 to prevent the establishment of the Asian hornet (*Vespa velutina*) in the UK (The Great Britain Non-native Species Secretariat 2015), and despite a number of colonisations this has, to date, been successful.

The National Bee Unit keeps a national record of honey bee hives, with 247,461 registered in 2017 across the UK (see section 1.2). However, registering is a voluntary process, and so there is only an incomplete picture of the number of hives, number of beekeepers, distribution of both, and movement of hives within years between apiaries and foraging/pollination sites. There is currently no regulation or management of honey bee hives that are moved into, or maintained within conservation areas, despite the potential threat they pose to wild bees (Fürst *et al.* 2014, Geldmann & Gonzalez-Varo 2018). As a result, how these factors relate to bee health remains unknown. However, a recent pan-European study that included data from England showed that the key factors driving honey bee health were beekeeper background and apicultural practices, indicating the need for further improvements in beekeeper training and surveillance (Jacques *et al.* 2017).

There is evidence from European countries that high densities of honey bees can have negative impacts on wild pollinators through competition, by depleting pollen and nectar resources or displacing wild pollinators in both crop fields (e.g. Lindstrom *et al.* 2016) and surrounding natural habitats (e.g. Magrach *et al.* 2017; Torné-Noguera *et al.* 2016). A systematic review of evidence on whether managed bees have negative impacts on wild bees (Mallinger *et al.* 2017) concluded that results on competition between wild and managed bees were highly variable, with 53% of 78 studies worldwide reporting negative effects on wild bees, while 28% reported

no effects and 19% reported mixed effects (varying with the bee species or variables examined). There is a current debate in the scientific literature about whether actions to support managed pollinators, such as promoting bee keeping and managed bee health, should or should not be included as part of pollinator conservation strategies (Colla and MacIvor 2017; Geldmann and Gonzalo-Caro 2018; Kleijn *et al.* 2018).

### 4.3.2 Bumble bees

Commercial bumble bees are used within glasshouses, in open and closed polytunnels, and in open field situations. However, there is no available information on how many commercial colonies of the native bumble bee subspecies *Bombus terrestris audax* are imported into the UK every year, nor of where or how they are used. For details of importation licensing please see section 1.2.2.

Commercial producers practice disease screening (Natural England 2009) and provide guidance on use and disposal of commercial colonies, to limit the release of new queens and males into the environment. We have no understanding of how effective these management strategies are – a study in Ireland suggested spread of diseases from commercial colonies into the surrounding bee assemblage (Murray *et al.* 2013), but this occurred prior to the introduction of the latest disease screening methodology. There has been no assessment of establishment by, or genetic introgression from commercial bumble bees into native populations in the UK. The commercial sale, cross-border transport, and use of native subspecies of bumble bees for pollination in England is currently largely governed by policy developed for honey bees, which is of arguable relevance (Council Directive 92/65/EEC, Commission Regulation (EU) No. 206/2010).

### 4.3.3 Solitary bees

At least two species of solitary bee (*Osmia bicornis, O. cornuta*) are currently commercially produced in Europe for pollination services in orchards (Sedivy & Dorn 2014). Successful and sustained production of these species requires management practices that control a range of parasites, predators, parasitoids, and pathogens (Sedivy & Dorn 2014). Recent work has demonstrated replication of so-called 'honey bee viruses' in *O. bicornis* and other solitary bees (Radzeviciute *et al.* 2017). As yet there is no management practice to control viruses in commercially produced solitary bees. The commercial sale, cross-border transport, and use of solitary bees for pollination is not covered by policy developed for honey bees and bumble bees (Directive 92/65). While imports would still be captured under the EU veterinary checks legislation (Directive 97/78), in the absence of any relevant conditions for health checks of solitary bees, this would have to be dealt with on a case by case basis. Consequently, we have no understanding of how the use of managed solitary bees may be impacting their health, or the health of wild pollinators with which they may come in contact.

### 4.3.4 Other pollinators

Pollinators such as hoverflies (syrphids) and green bottle flies are currently produced commercially for pest control (e.g., www.biobestgroup.com). Their potential as pollinators is growing (e.g., Hodgkiss *et al.* 2018). However, currently there is no specific policy or practice guidance for hoverflies on (i) how to regulate their import into England, or (ii) how to use them in ways that support their own health, as well as that of other wild or managed pollinators that might encounter them, which may be of concern given their potential role in the epidemiology of so-called 'bee viruses' (Bailes *et al.* 2018).

### 4.3.5 Knowledge gaps

- Greater understanding of the numbers, distribution, and health of managed honey bee hives across England.

- Record the import, sale, and distribution of managed bumble bee colonies, solitary bees and hoverflies.

- Determine the effectiveness of current management on numbers of honey bee hives, and effectiveness of different methods for the detection and treatment of honey bee diseases.

- Investigate the impact, if any, of managed honey bees, bumble bees, solitary bees, and hoverflies on wild pollinator populations.

- Develop targeted and appropriate policy for the use of solitary bees, hoverflies, and other flies as managed pollinators.

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