

Chapter 27

Conservation of bumblebees

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Summary

1. Declines in bumblebee species in the last 60 years are well documented in Europe, where they are primarily driven by habitat loss and declines in floral abundance and diversity, in turn driven by changing agricultural practices. Impacts of habitat degradation and fragmentation are likely to be compounded by the social nature of bumblebees and their largely monogamous breeding system which renders their effective population size low.
2. Recent studies suggest that surviving populations of some rare species consist of <30 breeding females, and such populations are susceptible to chance extinction events and inbreeding.
3. In North America, catastrophic declines of some bumblebee species since the 1990s are probably attributable to the accidental introduction of a non-native parasite from Europe, a result of global trade in domesticated bumblebee colonies used for pollination of greenhouse crops.
4. Given the importance of bumblebees as pollinators of crops and wildflowers, it is vital that steps be taken to prevent further declines. Suggested measures include tight regulation of commercial bumblebee use and targeted use of agri-environment schemes to enhance floristic diversity in agricultural landscapes.

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27.1 Introduction



Machair grassland on South Uist. © D. Goulson

The world bumblebee (*Bombus*) fauna consists of approximately 250 known species, largely confined to temperate, alpine and arctic zones of the northern hemisphere (Williams, 1994). There is mounting evidence that many bumblebee species have declined in recent decades, particularly in developed regions such as Western Europe and North America (reviewed in Goulson, 2003a; Kosior *et al.*, 2007, see also Thorp and Shepherd, 2005). In the UK, three of the 25 native species have gone extinct and a further eight species have undergone major range declines (Goulson, 2003a). The most severely affected species tend to be those with long tongues associated with deep perennial flowers (Goulson *et al.*, 2005). Similar patterns are evident in Europe. In a review of declines in bumblebees of 11 central and western European countries, Kosior *et al.* (2007) describe extinctions of 13 species in at least one country between 1950 and 2000. Four species (*B. armeniacus*, *B. cullumanus*, *B. serratissima*, *B. sidemii*) went extinct throughout the entire region.

A large number of wild plants are pollinated predominantly or exclusively by bumblebees, sometimes by particular species of bumblebee (Goulson, 2003a). Most bumblebees are generalist pollinators and most insect-pollinated plants use multiple pollinators (Waser *et al.*, 1996), so it could be argued that pollination networks are buffered against the loss of a few pollinator species. However, a recent study simulating the effects of removal of individual pollinators from pollination networks demonstrated that removal of highly linked pollinators such as bumblebees produced the greatest rate of decline in plant species diversity (Memmott *et al.*, 2004). Reduced pollination services can be particularly detrimental when plants are already scarce and threatened directly by the same changes in land use that threaten the bees (Goulson, 2003a; Goulson *et al.*, 2008).

Aside from the implications for conservation, there are good financial reasons for conserving bumblebees. The yields of many field, fruit and seed crops are enhanced by bumblebee visitation (Goulson, 2003c). For example, field beans in Europe are largely pollinated by longer tongued species such as *B. pascuorum* and *B. hortorum*, without which, yields are poor (Free and Williams, 1976). In the US there is an ongoing decline in managed honeybee populations due to disease, misuse of pesticides, loss of subsidies and dangers associated with invading Africanized honeybees (Kremen *et al.*, 2002). The value of crop pollination by honeybees in the US has been estimated at between 5 and 14 billion dollars per year, but beekeeping has diminished by around 50% over the last 50 years (Kremen *et al.*, 2002). This has given rise to concerns over the future of insect-pollinated crops such as cucumber, pumpkin, watermelon, blueberry and cranberry (Delaplane and Mayer, 2000; Richards, 2001; Kremen *et al.*, 2002). At sufficient densities, bumblebees pollinate many of these crops efficiently, often more so than honeybees (Stubbs and Drummond, 2001). However, the impoverished bumblebee communities often associated with agricultural landscapes may be insufficient to replace the services currently provided by honeybees.



Plate 27.1 *Bombus distinguendus*, one of the UK's rarest bumblebee species, feeding on red clover. © D. Goulson

27.2 Causes of bumblebee declines

Most researchers are convinced that the primary cause of bumblebee declines in Western Europe is the intensification of farming practices, particularly during the latter half of the 20th century (Goulson, 2003a, d). In the UK, a self-sufficiency drive in the wake of the Second World War led to a number of major changes. Permanent unimproved grassland was once highly valued for grazing and hay production but the development of cheap artificial fertilizers and new fast-growing grass varieties meant that farmers could improve productivity by ploughing up ancient grasslands. Hay meadows gave way to monocultures of grasses which are grazed or cut for silage. Between 1932 and 1984 over 90% of unimproved lowland grassland was lost in the UK (Howard *et al.*, 2003).

There is evidence to suggest that bumblebee forage plants have suffered disproportionate declines. A recent study in the UK found that of 97 preferred bumblebee forage species, 69 (71%) have suffered range restrictions, and 74 (76%) have declined in abundance over the past 80 years, exceeding declines of non-forage species (Carvell *et al.*, 2006). Leguminous crops (notably clovers, *Trifolium* spp.) used to be an important part of crop rotations in much of Europe, and these are highly preferred food sources, particularly for long-tongued bumblebee species (Goulson *et al.*, 2005). Since the introduction of cheap artificial fertilizers, rotations involving legumes have been almost entirely abandoned, and it has been argued that this is one of the primary factors driving the decline of long-tongued bumblebees (Rasmont and Mersch, 1988; Goulson and Darvill, 2004).



Plate 27.2 Machair grassland on South Uist, a rare surviving example of species-rich grassland. © D. Goulson



Plate 27.3 *Bombus hortorum*, the only long-tongued bumblebee that remains common in western Europe. © B. Darvill

Uncropped areas of farmland, such as hedgerows, field margins and borders of streams may provide flowers throughout the season, and therefore support greater numbers of foraging bumblebees than cultivated areas (Banaszak, 1992; Mänd *et al.*, 2002). However, these areas will be adequate only if there are enough of them, and if they have not been degraded by drift of herbicides and fertilizers. Insufficient flower-rich uncropped areas may lead to gaps in the succession of flowering plants during which bumblebee colonies may starve and die. With a decline in bees, the plants that they pollinate set less seed, resulting in less forage for the bees in subsequent years (Ockinger and Smith, 2007). The process by which mutually-dependent species drive each other to extinction is known as an ‘extinction vortex’. We do not as yet know whether this process is really occurring, but it is clear that farmland provides less food for bees than it once would have done.

In addition to floral resources, bumblebees need suitable nesting sites, the precise requirements for which vary between species (Kells and Goulson, 2003). The carder bees (*Thoracobombus*) such as *B. pascuorum* tend to nest in dense grassy tussocks while other species such as *B. terrestris* nest underground in cavities. Both groups often use abandoned rodent nests. The loss of hedgerows and of unimproved pastures is likely to have reduced availability of nest sites for both above and below-ground nesting bumblebee species (Banaszak, 1992). Those species that nest above ground frequently have their nests destroyed by farm machinery, particularly by cutting for hay or silage. The scarcity of weeds and field-margin flowers on modern intensive farms means that there are less seeds, and therefore less food for voles and mice. Lower populations of these mammals will lead to fewer nest sites for both above and below-ground nesting bumblebee species.

In California, even on organically managed farms, the presence of bumblebees, in this case *B. vosnesenskii* and *B. californicus*, depends on proximity to areas of natural habitat in which the bees can nest (Kremen *et al.*, 2002). A study in Sweden found that field boundaries within 100m of a semi-natural grassland area contained a greater abundance and diversity of foraging bumblebees than similar sites > 1 km from such habitat. However, bumblebee abundance was found to be significantly lower in the semi-natural grasslands themselves suggesting that these sites were used primarily for nesting (Ockinger and Smith, 2007). Similarly, Greenleaf and Kremen (2006) found that tomato fields in northern California obtain high visitation rates from *B. vosnesenskii* only when they were positioned within 300m of a patch of natural habitat and if at least 40% of the land within a 2,100m radius of the farm was natural habitat.

27.2.1 Impacts of alien bees

It has been argued that the most immediate threat to bumblebees in the US, unlike in Western Europe, is the spread of disease due to widespread trafficking of commercial bumblebee hives (Thorp and Shepherd, 2005). Commercial bumblebee hives are used for greenhouse pollination all over the world, including; Israel, Korea, Japan, North America and Europe (Goulson, 2003b). In the US, colonies of *B. impatiens* and *B. occidentalis* have been commercially reared since the early 1990s for the pollination of greenhouse crops such as tomatoes (Whittington and Winston, 2004) and sweet peppers (Shipp *et al.*, 1994). These colonies have been found to have a greater parasite load than wild colonies with an elevated prevalence of the bumblebee specific protozoan pathogens *Crithidia bombi* and *Nosema bombi*, and of the tracheal mite *Locustacarus buchneri* (Colla *et al.*, 2006). These parasites have detrimental effects on colony survival and reproduction and/or the foraging efficiency of individual workers (Brown *et al.*, 2003; Gegear *et al.*, 2005; Otterstatter *et al.*, 2005).

There is little doubt that there are greenhouse escapes. In Japan, feral colonies of the non-native *B. terrestris* are now common (Inari *et al.*, 2005), and a recent study in Canada found that 73% of pollen carried by workers returning to commercial colonies originated from plants outside the greenhouse (Whittington *et al.*, 2004). Consequently, there is a high likelihood of interaction between wild and commercially reared bees at flowers, providing conditions for 'pathogen spillover' from the commercial

population to wild populations. Significant increases in the prevalence of *C. bombi* and *N. bombi* have been found in wild bumblebee populations near to commercial greenhouses, compared to wild populations elsewhere (Colla *et al.*, 2006). In 1998, a *N. bombi* outbreak was reported in bumblebee production facilities in North America. This was thought to be a result of the importation of infected European *B. terrestris* colonies into Mexico in 1995 and 1996 (Winter *et al.*, 2006). Similarly, *C. bombi* has only been detected in the US since use of commercially reared bumblebees began and it is suspected that this parasite is not native to the US (Winter *et al.*, 2006). The introduction may have occurred as a result of the shipment of queens of *B. occidentalis* to Europe for commercial rearing before re-importation into the US in the early 1990s (Colla *et al.*, 2006; Winter *et al.*, 2006). Exposure to a non-native pathogen is a likely cause of the catastrophic declines in *B. terricola*, *B. affinis*, *B. franklini* and *B. occidentalis* (Whittington and Winston, 2004; Thorp, 2005; Thorp and Shepherd, 2005). However, we have a very poor understanding of the relative susceptibilities of bumblebee species to parasites, or of the natural distributions of these parasites, and work is urgently needed in this area. It is clear that tight controls are needed on transport of domesticated bumblebee hives.

There are other dangers associated with commercial trafficking of bumblebees. *B. terrestris* is now naturalized in Japan and there are concerns regarding possible competitive effects of this species on native bumblebees. Studies have shown that *B. terrestris* has four times the reproductive output of native species (Matsumara *et al.*, 2004) and that there are considerable overlaps in forage use and timing of foraging (Inari *et al.*, 2005). There are similar concerns in the UK where around 10,000 colonies of a south-eastern European subspecies *Bombus terrestris dalmitinus* are imported each year (Ings *et al.*, 2006). Britain has an endemic subspecies of this bee, *Bombus terrestris audax*. Evidence suggests that there are dangers to the British subspecies in the form of parasite transmission (Ings *et al.*, 2005) or out-



Plate 27.4 *Bombus muscorum*, has declined precipitously in England but healthy populations survive in the Western Isles. © D. Goulson

competition, particularly since the introduced subspecies has superior foraging efficiency and reproductive rate (Ings *et al.*, 2006). Also, *B. terrestris dalmitinus* and *B. terrestris audax* readily interbreed, so the native subspecies could be lost through introgression (Ings *et al.*, 2005).

In addition to non-native bumblebee species, native pollinator communities in many parts of the globe also have to contend with other introduced bee species, most notably honeybees (*Apis mellifera*). These natives of Europe, Africa and the Middle East have been introduced by man to almost every country in the world. Their impacts are reviewed by Goulson (2003b). Recent studies suggest that honeybees can have negative effects on bumblebees. Walther-Hellwig *et al.* (2006) found that short-tongued bumblebees avoided areas of forage close to honeybee hives, while carder (*Thoracobombus*) bumblebees switched to foraging later in the day and were displaced from their preferred foodplant. Thomson (2004) experimentally introduced honeybees and found that proximity to hives significantly reduced the foraging rates and reproductive success of *B. occidentalis* colonies. In eastern US, Thomson (2006) found a strong overlap between the foraging preferences of the two species, which peaked at the end of the season when floral resources were scarce, corresponding with a negative relationship between honeybee and bumblebee abundance.

There is increasing evidence that the spread of natural enemies of bumblebee colonies is being aided by honeybees. Honeybees can act as vectors for the bumblebee specific *C. bombi* via flowers (Ruiz-Gonzalez and Brown, 2006). The African honeybee parasite *Aethina tumida* (small hive beetle) recently invaded North America, Egypt and Australia, and attacks *B. impatiens* colonies where it causes considerable damage (Spiewok and Neumann, 2006). Deformed wing virus, a viral honeybee pathogen, has been found in commercial colonies of *B. terrestris*, transmitted between the two species as a result of the practice of placing honeybees with queens to induce colony founding (Genersch *et al.*, 2006). However, it has also been found in a wild colony of *B. pascuorum* which had been robbing a managed honeybee hive (Genersch *et al.*, 2006). This virus appears to have higher virulence to bumblebees than to honeybees and the findings raise important questions about transmission and cross-infectivity between bumblebees and honeybees.

27.2.2 Bumblebee population structure

As a consequence of the various factors discussed above, populations of a number of bumblebee species have become increasingly small, fragmented and separated from one another by large distances. In the UK, where distributions are best known, declines appear to have followed a characteristic pattern. The last bumblebee species to disappear from the UK (*B. subterraneus*), was once widespread across southern England, but declined rapidly in the years after the Second World War. By the 1980s the few remaining populations were small and isolated, surviving on habitat islands (nature reserves) that had escaped agricultural intensification. However, these populations subsequently disappeared despite the apparent suitability and protected status of the remaining habitat (Goulson, 2003a). The species was last recorded at Dungeness National Nature Reserve in 1988. Several other UK species such as

B. distinguendus and *B. sylvarum* are in the late stages of a similar process, and are likely to go extinct in the near future. Why do isolated populations go extinct? Understanding the consequences of the fragmentation of remnant populations of bumblebees is of great importance to conservationists, given the current distributions of many rare species.

Small populations of all taxa are inherently more vulnerable to local extinctions due to environmental and demographic stochasticity (Frankham *et al.*, 2002). If these populations form part of a broader metapopulation then regional extinctions can be balanced by subsequent recolonization, but if fragmentation is severe then extinct patches may never be repopulated. In addition, a functioning metapopulation ensures that dispersal maintains genetic cohesion. However, if habitat fragmentation results in the isolation of populations, then they may face an additional extinction threat through inbreeding (Frankham *et al.*, 2002). There are a number of reasons why bumblebees may be particularly badly affected by habitat fragmentation. It is the effective population size (N_e), rather than the census population size (N_c) which determines the rate of genetic drift in a population, and N_e may be several orders of magnitude lower than N_c . In bumblebees, as in many other social insects, N_e depends on the number of successful colonies. The N_e contributed by an individual colony depends on the number of egg-laying queens and the number of males they have mated with, but (unlike many other hymenopterans) bumblebee colonies are all founded by a single queen, and the vast majority of species are monoandrous (Estoup *et al.*, 1995; Schmid-Hempel and Schmid-Hempel, 2000). It seems therefore that population sizes of bumblebees may be low, even relative to other social insects, making them particularly susceptible to the loss of genetic diversity.

Given the potentially serious consequences of inbreeding in bumblebees, it is essential that we understand its prevalence within wild bumblebee populations. Until recently, studying the population genetics of rare bee species was extremely difficult, as lethal sampling was necessary. Work in this area was greatly aided by the development of a non-lethal DNA sampling technique (Holehouse *et al.*, 2003), and this has recently been applied to studies of fragmented populations of rare species: *B. muscorum* (Darvill *et al.*, 2006), *B. sylvarum* (Ellis *et al.*, 2006) and *B. distinguendus* (Bourke and Hammond, 2002). All three studies found significant population structuring. For example in *B. muscorum*, all populations >10 km apart were significantly differentiated, as were some populations just 3km apart, suggesting that this species has very limited dispersal abilities. Ellis *et al.* (2006) used microsatellite markers to group workers into sisterhoods and so estimated the number of colonies (and hence N_e) in populations of *B. sylvarum*, a species which is highly endangered in the UK. Estimates of N_e were very low (range 21-72) suggesting that these populations are very vulnerable to loss of genetic diversity through drift. In all rare species studied to date, genetic diversity (allelic richness and heterozygosity) is low compared to common species (Table 27.1). If fragmented populations of rare bumblebee species are suffering from reduced fitness through inbreeding then we must take steps to conserve what genetic diversity remains. Management strategies in vertebrates routinely consider genetic factors, and we may need to adopt similar measures in the management of rare bumblebee populations.

Table 27.1 Genetic diversity estimates for populations of a number of *Bombus* species (mean \pm S.E.). Reviewed in Goulson *et al.* (2008).

Species	Population	Sample Size	Allelic Richness	H_E
<i>B. ignitus</i>	Beijing, China	33	12.2 \pm 1.53*	0.85 \pm 0.02
<i>B. ignitus</i>	Nagano, Japan	26	8.22 \pm 0.72*	0.83 \pm 0.03
<i>B. pascuorum</i>	Landford, UK	183	6.22 \pm 1.19*	0.52 \pm 0.15
<i>B. pascuorum</i>	Rothamsted, UK	125	5.71 \pm 1.01	0.52 \pm 0.11
<i>B. pascuorum</i>	Continental Europe	22.7 average	5.49 \pm 0.16*	0.56 \pm 0.01
<i>B. terrestris</i>	Continental Europe	37.5 average	5.96 \pm 0.12*	0.61 \pm 0.01
<i>B. lucorum</i>	Bern, Switzerland	40	7.00 \pm 2.00*	0.60 \pm 0.12
<i>B. hypnorum</i>	Various, Sweden	10	6.75 \pm 1.03*	0.72 \pm 0.14†
<i>B. sylvarum</i>	Southern UK	25.6 average	3.12 \pm 0.10	0.39 \pm 0.02
<i>B. sylvarum</i>	Epenede, France	10	4.00 \pm 0.85	0.53 \pm 0.09
<i>B. muscorum</i>	Outer Hebrides, UK	43.8 average	3.22 \pm 0.12	0.39 \pm 0.01
<i>B. muscorum</i>	Inner Hebrides, UK	62.7 average	3.21 \pm 0.07	0.47 \pm 0.01
<i>B. muscorum</i>	Southern UK	35.5 average	4.01 \pm 0.06	0.51 \pm 0.01
<i>B. distinguendus</i>	Scotland, UK	7.75 average	2.63 \pm 0.23*	0.42 \pm 0.01 †

* allelic richness was not available, and the average number of alleles per locus is presented. Allelic richness is a normalized measure which takes account of differing sample sizes to give a comparable figure for all populations.

† Expected heterozygosity (H_E) was not available, and observed heterozygosity is given. These measures are expected to be very similar for populations that are in Hardy-Weinberg equilibrium.

27.3 Conservation measures

A major cause of bumblebee declines is undoubtedly loss of habitat to intensive farming. However, there are moves to reverse this trend in Europe and North America where there is a growing emphasis on combining the goals of agriculture and conservation (Ovenden *et al.*, 1998; Kleijn and Sutherland, 2003). Subsidies are currently available in many countries for agri-environment schemes that promote biodiversity, including replanting of hedgerows, leaving land fallow, sowing wildflower strips, and restoring flower-rich grassland. Most of the management options promote floral abundance and diversity. It has been found that a 6m wide field margin kept free of crops and agrochemicals may contain six times as many flowering plants and ten times as many flowers than the equivalent cropped area (Kells *et al.*, 2001). The effects of UK field margin management options on bumblebee communities have been the focus of many studies in recent years.

The most valuable form of field margin management for bumblebees has been found to be the sowing of either wildflowers or a pollen and nectar mix consisting of agricultural cultivars of legume species (Carreck and Williams, 2002; Carvell *et al.*, 2004, 2007; Pywell *et al.*, 2005, 2006). Carvell *et al.* (2007) found that the pollen and nectar mixture produced the highest flower abundance with a succession of forage plants flowering over the 3-year trial period. The wildflower mixture produced few flowers in the first year but flower abundance increased over the three years as the mixture became established. Both treatments led to an increase in bumblebee species richness and abundance, and in the third year the wildflower mix was as valuable as the pollen and nectar mixture. Once established, the wildflower mix should persist for up to ten years, while the agricultural cultivars in the pollen and nectar mix are likely to need re-sowing within five years (Pywell *et al.*, 2002).



Plate 27.5 Overgrazed and degraded machair grassland with few flowers. The contrast with Plate 27.2 is striking. © N. Redpath

Bumblebees not only require a suitable source of forage, but also nest and hibernation sites. A popular agri-environment scheme in the UK is the sowing of field margins with tussocky grasses (Pywell *et al.*, 2006). These habitats attract the small mammals whose abandoned holes are used by bumblebees for nest sites (Svensson *et al.*, 2000), so it is likely that this form of management is of value to bumblebees. Carvell *et al.* (2004) found that field margins sown with a 'split' treatment consisting of a mixture of tussocky grasses and wildflower mix attracted almost as many bumblebees as margins sown solely with the wildflower seed suggesting that it is possible to provide both forage and nesting habitat in small areas.

27.4 Conclusions

Widespread declines of bumblebee species threaten pollination services to both wildflowers and crops. It is clear from studies of population structure that most bumblebee species cannot be conserved by managing small protected 'islands' of habitat within a 'sea' of unsuitable, intensively farmed land. Large areas of suitable habitat are needed to support viable populations in the long term. Also, studies of foraging range indicate that bumblebees exploit forage patches at a landscape scale, so that the scale of management must be appropriate. An integrated approach across large areas or several farms is more likely to succeed than localized efforts. Where small, isolated populations of rare species remain in habitat fragments, targeting the adjacent farms for uptake of suitable agri-environment schemes could increase the population size and so reduce the likelihood of stochastic extinction events and inbreeding. Similarly, such schemes could be used to provide linkage between habitat islands.

Unimproved flower-rich grassland is one of the most important habitats for bumblebees, but has been largely lost to agriculture in Western Europe and North America. Restoration of areas of this habitat will boost bumblebee populations and has been shown to provide improved pollination services on nearby farmed land. Substantial benefits could also be obtained by reintroducing clover (e.g. *Trifolium pratense*) ley crops into rotations, since this is a key forage source for many declining bumblebee species. This would also reduce dependency on artificial fertilizers.

In the US, recent declines in several bumblebee species have been linked to increases in the commercialization of bumblebees for greenhouse pollination and associated introductions of parasites. There are already restrictions in place on the importation and movement of bumblebees in Canada, Mexico and the US but there are calls for increased restrictions on transportation of bees and for stricter quarantine and monitoring systems (Winter *et al.*, 2006). Similar systems are urgently needed in Europe.

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