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Review article

Assessing the exposure and toxicity of pesticides to bumblebees (*Bombus* sp.)

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Abstract – Many of the reported pesticide incidents involving honeybees probably also result in mortality of bumblebees and, together with a reduction in suitable habitat, these have resulted in the decline in bumblebees in the UK over the last 20 years. Applications of sprays, e.g. pyrethroids, to flowering crops or weeds at times when honeybees are less active are likely to result in unreported bumblebee deaths. There is a need to protect foraging bumblebees from direct overspray during the early morning and late evening when pesticides which are repellent but highly toxic are applied, i.e. pyrethroids. Of particular concern are those pesticides applied when queens are emerging and establishing colonies, e.g. March/April, when colonies may be significantly impacted by the loss of a small number of workers or the queen. This is a problem which cannot readily be addressed by risk management measures due to differing foraging profiles of honeybees and bumblebees but does need to be taken into account in risk assessment and the development of more selective compounds.

***Bombus* / pesticides / exposure / toxicity / risk assessment**

1. INTRODUCTION

There has been a severe decline in the abundance of bumblebees (*Bombus* sp., Apidae) in the last thirty years, particularly in southern Britain, and it is possible that this is due in part to the use of certain pesticides. Bumblebees are important pollinators of some crops and many wildflowers and are considered 'beneficial' insects. Many pesticides in current use, such as pyrethroids, are known to be more toxic at lower temperatures (Inglesfield, 1989) and are applied in the early morning or late evening. Therefore these pesticides may be a greater threat to foraging bumblebees, which fly at lower temperatures than honeybees. Deaths of bumblebees due to pesticides are unlikely to be reported, since these bees are not kept domestically and, because their colonies are

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relatively small, they will die in small numbers.

All bumblebee species form colonies which are small in comparison to honeybee colonies. A colony of several hundred workers is considered large in bumblebees (Alford, 1975) compared with a full colony size of around 30000 individuals, for honeybees (Seeley, 1985). The life cycle of bumblebees in temperate regions differs from that of honeybees in that only the queens over-winter. The rest of the colony – workers and males – survive only for a single season. Bumblebee species differ considerably in the details of the life cycle, having different colony life times and seasonalities. Mated queens emerge from February to May (in the northern hemisphere) depending on species, feeding and establishing a new colony, collecting pollen and nectar to feed their first batch of worker larvae, and feeding themselves. Often the queen works alone for more than a month and this is a time when bumblebees are much more vulnerable than honeybees. In addition, as they have smaller colonies even when fully established, a single bumblebee worker is more important to the survival of the colony than a single honeybee worker. This review evaluates the data available on pesticide exposure and toxicity in bumblebees and identifies possible risk assessment approaches.

2. ASSESSING PESTICIDE EXPOSURE

2.1. Crops

A review of literature was carried out to collate currently available information on the use of crop plants by bumblebees (Thompson and Hunt, 1999). All insect-pollinated crops known to be grown in the UK were included and cereal crops were considered as a single group. Of 59 crops considered (not including cereals), bumblebees are known to visit 43 (73%) of them and are likely to visit a further 13. Bumblebees were consistently faster foragers (visited more flowers per minute) than honeybees on all except on three of the crops listed (Sunflower (*Helianthus annuus)*, cherries/plums *(Prunus* sp.), blackberry (*Rubus fruticosus*)). No records of bumblebees collecting honeydew from aphids on cereals crops could be found. However, *Bombus lucorum* and other species have been reported to collect honeydew, usually on trees, in Russia, Finland, USA and the UK (Bishop, 1994; Brian, 1957; Teras, 1985). It is reasonable to suggest that bumblebees actually use the majority of our insect-pollinated crop plants to a large extent and they are therefore likely to be exposed to many pesticide applications.

2.2. Wild flowers

Bumblebees will also be exposed to pesticides if they are foraging on wild flowers that grow under crops or in field margins (Thompson and Hunt, 1999). Wild flowers associated with crops were divided into two groups – weeds (mostly annuals) which grow in ploughed fields with crops; and field margin species (mostly perennials) which grow in unploughed field boundaries and hedgerows. Bumblebees using the former group, arable weeds, are likely to be more exposed to pesticides than bumblebees foraging on field margins, because farmers should be trying to avoid spraying the field margins and hedgerows with pesticides.

The arable weed flora consists of plants growing underneath crops and often up to the field edge. Due to the annual disturbance of ploughing, perennial plants are unable to establish here, with the exception of those which can regenerate from broken root fragments such as field bindweed (*Convolvulus arvensis*). The flora is therefore made up predominantly of annual species, most of which are not favoured by bumblebees. However, any bees foraging on these species will be directly exposed to pesticide sprays. Four of the weed species – field pansy (*Viola*

arvensis), field bindweed (*Convolvulus arvensis*), deadnettles (*Lamium* sp.), and common poppy (*Papaver rhoeas*) exhibit features common to bumblebee pollinated flowers (Proctor et al., 1996). Two of these – red dead-nettles (*Lamium purpurem*) and field bindweed (*Convolvulus arvensis*) – are considered to be important to certain types of bumblebee in the seasonal succession of forage plants. This suggests that bumblebees will be exposed to pesticides even when they are not foraging on the crop.

In Britain, the flora of field margins and hedgerows is much more variable than the arable weed flora (Barr et al., 1996; Mountford et al., 1994). Fussell and Corbet's (1992) national survey of flowers used by bumblebees was used as a basis for compiling a list of field margin and hedgerow flowers. Thirty-one plant species were identified, all of which are common in the field margin/hedgerow in at least some parts of the country (Stace, 1997). Woody species that make up hedges, such as hawthorn, were not included as the flowers are less likely to be exposed to pesticides. All those identified were either perennial or biennial, with the exception of borage and some species of vetch and geranium, and all are considered important forage for bumblebees. *Lamium album*, white deadnettle, is of particular importance. It is an early flowering species and it is used extensively in the early Spring by foraging queens of long-tongued species (Fussell and Corbet, 1992; Prys-Jones, 1982).

2.3. Seasonal and diel foraging patterns of bumblebees

In general, the bumblebee season runs from mid-March to mid-October in Britain, with a peak in numbers during the summer. The timing of queens emerging from hibernation correlates with temperature (Goodwin, 1995; Prys-Jones, 1982). The order in which the species emerge from hibernation is relatively consistent and the two long-tongued species (*B. pascuorum* and *B. hortorum*) are the last species to emerge. This means that foraging queens of these species in the process of founding colonies can be vulnerable to the effects of pesticides later in the year than those of other species.

The foraging and flying activity of bumblebees during the day has been recorded by many authors. Normally, the pattern observed is that the number of foragers peaks in the early morning and evening, with a drop in numbers in the middle of the day (Plowright and Laverty, 1984; Alford, 1975). Bumblebees also tend to start foraging earlier in the day than honeybees, and finish later in the evening (Fussell and Corbet, 1991; Corbet et al., 1993). This is very different from the activity pattern for honeybees, in which the number of foragers peaks in the middle of the day (Fig. 1). The combination of two factors is thought to combine to account for the difference in foraging activity between bumblebees and honeybees – the effects of ambient temperature and the effects of exploitative competition with honeybees.

2.4. Likelihood of exposure of queens and workers to pesticides

There are a number of classes of insecticide applied to crops during the periods when queens are emerging and establishing colonies (Thompson and Hunt, 1999). Amongst these are the pyrethroid insecticides which are often applied to flowering crops during early morning or late evening. Therefore, pyrethroids can be applied to oilseed rape crops in full flower at times of day when, although honeybees are less active, bumblebee species are at their most active.

Although the application of insecticides to non-flowering crops is not of immediate concern there are likely to be flowering weeds around many of these crops which are attractive to bumblebees. Generally

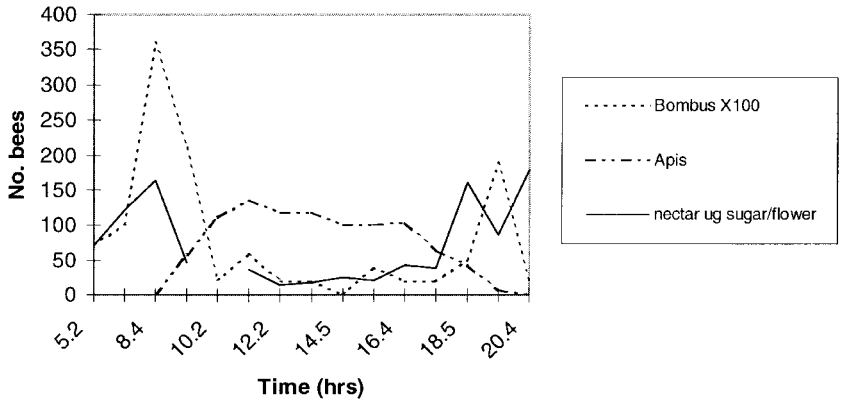

Figure 1. Comparison of diel foraging activity of honeybees and bumblebees.

queens emerge between February and mid-June depending on species and during this period many of the common weed species are in flower. Flowering weeds are unlikely to be visited by significant numbers of honeybees and therefore are likely to be overlooked by spray operators. Therefore the presence of flowering weeds in and around agricultural crops is probably important in determining the exposure of bumblebees.

3. ASSESSING TOXICITY

3.1. Contact and oral toxicity

A number of methods for testing the toxicity of pesticides to bumblebees (Tab. I) have evolved over the last few years based on the established methods for honeybee toxicity testing (OECD, 1998a; OECD, 1998b). By far the majority of tests have been developed using *B. terrestris* which is the species used commercially for pollination, e.g. in glasshouses, and is therefore readily available.

Table II shows the contact and oral toxicity of a range of pesticides to honeybee and bumblebee species both in terms of per bee and per g bee. There are very limited data for bumblebee species other than *B. terrestris* although the data available shows the toxicity on a weight basis to be similar. It can be seen that generally the toxicity of the pesticides, for which data are available, are generally lower to bumblebees than honeybees when expressed on a weight basis. However, it should also be remembered that the data are limited in terms of number and type of insecticide. There is a need to increase the amount of toxicity data available for bumblebees in order to support the assumption that they are less susceptible than honeybees.

3.2. Brood effects

A small number of studies have established methods to assess the effects of insect growth regulators (IGR) on bumblebee brood. The structure of the nest makes assessments of effects on larvae more complicated than for honeybees. (De Wael et al., 1995) evaluated the effects of IGRs using photographs of the colony to evaluate gross effects. Thompson and Barrett (in press) evaluated the use of a novel IGR on tomatoes on the viability of bumblebee colonies (approx. 200 workers) in glasshouses. The number of dead larvae ejected from the colony were assessed and compared with

Table I. Details of contact and oral toxicity testing methods for bumblebees.

Contact	Oral
1 µl acetone dissolved pesticide formulation, ventral thorax 2nd-3rd pairs of legs, 10 bees per concentration, 5 concentrations, 24 h mortality. Link between bumblebee size and LD_{50}, linear regression (Van der Steen et al., 1996)	Active ingredient dissolved in 50% sucrose, individually fed with calibrated pipettes, kept isolated, probit analysis mortality at 24, 48, 72 h, LD_{50} related to size of bee (Drescher and Geusen-Pfister, 1991)
(Sublethal test) Anaesthetised with CO_2, 1 µl drop formulated pesticide in acetone on thorax (Tasei et al., 1994)	Collected and kept as contact, fed on sucrose containing pesticide, treatment 4 groups of 8 workers, mortality and uptake checked daily (corrected for evaporation) (Tasei et al., 1994)
30 s CO_2, 1 µl drop formulated pesticide in acetone, controls acetone alone, applied to thorax, 8–10 bees per box 1 dm^3 fed 35% sucrose. 20 °C in dark, mortality checked daily (Tasei et al., 1994)	30 µl formulated pesticide dissolved in 50% sucrose, offered to individuals in micropipettes... then 72% sucrose ad libitum, kept in transparent cups 20 °C, 55% rel humidity, 4–6 doses, 30 bees per dose (Gretenkord and Drescher, 1993)
As honeybee test (Schaefer et al., 1993)	Formulated pesticide dissolved in 30% sucrose, fed to 10 bees of comparable body weight for 24 h (Schaefer et al., 1993)
Mortality in control ≤ 10%, mean weight of bees determined, anaesthetised for as short a time as possible, test substance dissolved in acetone, bees kept in dark at 25 ± 2 °C, 30 bees per concentration, 5 concentrations test substance, 2 replicates in time preceeded by range finding test, 1 µl test solution pipetted on ventral part of thorax between 2nd and 3rd pairs of legs, bees housed together by dose and fed sucrose solution ad libitum, mortality recorded 24, 48, 72 h. Toxic reference 40% dimethoate or 25% parathion 3 concentrations and acetone control, LD_{50} µg/bee or µg/g bee (Van der Steen et al., 1996)	30 bees per dose, mortality in control ≤ 10%, bees individually caged for dosing, mean weight of bees determined, deprived of food 2–3 h before dosing, not anaesthetised with CO_2, pesticide dissolved in sucrose, kept in dark at 25 ± 2 °C, 5 concentrations test substance, 2 replicates in time preceded b y range finding, 10 µl test solution fed so cannot be contaminated, 2 h dosing period, after dosing bees housed together by dose, and fed sucrose ad libitum, mortality recorded, 24, 48 and 72 h. Toxic reference 40% dimethoate or 20% parathion, 3 concentrations and control, LD_{50} µg/bee or µg/g bee (Van der Steen et al., 1996)

the results from control and diflubenzuron treated glasshouses (Fig. 2). This shows the increase in larval mortality over time in the IGR treated colonies with maximum levels reached by 9 days after application in the diflubenzuron treated colony and 15 days in the novel IGR treated colonies.

3.3. Sublethal effects

Table III shows the reported sublethal effects of pesticides on honeybees and bumblebees. Although there are a number of other sub-lethal effects which may not themselves cause concern, e.g. repellency and

Table II. Contact and oral toxicity of a range of pesticides to honeybee and bumblebee species.

Pesticide	Contact LD_{50} *A. mellifera*		Oral LD_{50} *A. mellifera*		Contact LD_{50} *B. terrestris*		Oral LD_{50} *B. terrestris*		Contact LD_{50} *B. lucorum*	Contact LD_{50} *B. agrorum* (*pascuorum*)		Oral LD_{50} *B. lapidarius*
	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg ai/bee	µg/g bee[1]	µg ai/bee
Phosalone	8.9[f]	89					60 (24 h Rubitox)[a]	286				
Pirimicarb	> 54[f]	> 540					8.5 (24 h Pirimor Granulat)[a]	40				
Oxydemeton methyl	0.54[f]	5.4					0.75 (24 h Metasystox R)[a]	3.6				
Deltamethrin	0.05 (24 h)[d]	0.5			0.9 (48 h Decis CE in acetone)[e]	4.3	0.6 (24 h Decis)[a]	2.7				
Acephate			0.2 (24 h)[b]	2.0			135.5 (24 h)[b], 3.93 (72 h)[b]	645 19				
Methomyl			0.08 (24 h)[b]	0.8			3.2 (24 h)[b], 2.6 (72 h)[b]	15 12				2.78 (24 h)[b], 2.4 (48 h)[b], 2.18 (72 h)[b]
Dimethoate	0.4 (24 h)[c] 0.12 (24 and 48 h)[f] 0.1 (24 h)[g]	4.0 1.2 1.0	0.12 (24 and 48 h)[h]	1.2	4.1–13 (24 h)[c] 4.8 (24–72 h)[h]	19–62 23	4.7 (24–72 h)[h]	22	workers 2–5 (24 h)[g] queens 5–20 (24 h)[g]	workers 0.5–2 (24 h)[g] queens 1–5 (24 h)[g]	4–17	
Alpha cypermethrin	0.03 (24 h)[d] 0.05 (24 h)[f]	0.3 0.5	0.06 (24 h)[d]	0.6	0.17 (24 h)[h] 0.15 (72 h)[h]	0.81 0.71	0.52 (24 h)[f] 0.36 (72 h)[f]	2.5 1.7				
Permethrin	0.1 (24 h)[d]	1.0	0.03 (24 h)[d]	0.3	0.81 (24 h)[h] 0.82 (72 h)[h]	3.9 3.9						

Table II. (Continued).

Pesticide	Contact LD_{50} *A. mellifera*		Oral LD_{50} *A. mellifera*		Contact LD_{50} *B. terrestris*		Oral LD_{50} *B. terrestris*		Contact LD_{50} *B. lucorum*	Contact LD_{50} *B. agrorum* (*pascuorum*)		Oral LD_{50} *B. lapidarius*
	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg/g bee[1]	µg ai/bee	µg ai/bee	µg/g bee[1]	µg ai/bee
Chlorpyrifos	0.059 (24 h)[f]	0.59			2.39 (24 h)[h] 1.58 (72 h)[h]	11.4 7.5						
Demeton S Methyl	0.26 (24 h)[f]	2.6			3.27 (24 h)[h] 2.68 (72 h)[h]	15.6 13						
Demeton methyl	0.5 (24 h)[g]	5.0							workers 1–2 (24 h)[g] queens 6–24 (24 h)[g]	workers 1–3 (24 h)[g] 8–25 queens 10–24 (24 h)[g]		
Disulfoton	5.0 (24 h)[g]	50							workers 2–10 (24 h)[g] queens > 40 (24 h)[g]	workers 1–4 (24 h)[g] 8–33 queens 5–10 (24 h)[g]		
Phorate	0.3 (24 h)[g]	3.0							workers 1–2 (24 h)[g] queens 6–23 (24 h)[g]	workers 1–2 (24 h)[g] 8–17 queens 1–5 (24 h)[g]		

[1] Calculated average based on *A. mellifera* = 0.10 g; *B. Terrestris* = 0.21 g, *B. pascuorum* = 0.12 g; [a] (Gretenkord and Drescher, 1993); [b] (Drescher and Geusen-Pfister, 1991), [c] (Van der Steen et al., 1996); [d] (Inglesfield, 1989); [e] (Tasei et al., 1994); [f] (Greig-Smith et al., 1994), [g] (Stevenson and Racey, 1966); [h] unpublished NBU data.

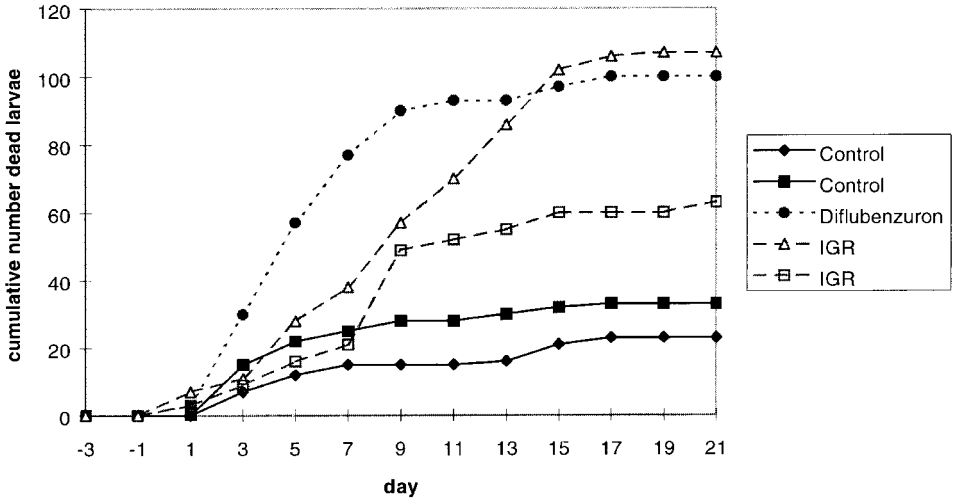

Figure 2. Number of dead larvae in treated glasshouses from 3 days before to 21 days after the first spray application of diflubenzuron or a novel IGR (applications were made on days 0 and 10) (Thompson and Barrett, 2000).

decreased house-cleaning, effects such as disruption of homing flights and decreased longevity may result in significant impacts on colonies. There is a need to account for these, as well as mortality, in pesticide risk assessment.

It appears at present that the repellency observed in honeybees exposed to pyrethroid insecticide treated crops also occurs in bumblebees. However, such repellency will be less effective in reducing the risk associated with these insecticides if bees are foraging on the crop at the time of application (i.e. they are oversprayed directly) or the pyrethroid toxicity is increased by application in a tank mix with EBI fungicides (Thompson, 1996).

It is apparent that further work is required to understand the sublethal impact of pyrethroid exposure on bumblebees. Although these pesticides are repellent, initial exposure of bees may seriously affect their ability to forage or return to the nest. In honeybees this has been observed as total loss of flying bees from a colony. Colonies of honeybees may be able to recover from such a loss through replacement from emerging brood at the height of the season. However, the far smaller size of bumblebee colonies results in such a loss having a potentially far greater impact.

3.4. Size variation and pesticide toxicity

The mass of an individual bee varies greatly according to larval nutrition, crop load and pollen load. Size variation within and between different colonies of the same species can be significantly greater than variation between species (Van der Steen et al., 1996). In general, queens are much larger than workers although there can be considerable size overlap (Prys-Jones, 1982). There are some generalisations which can be made, however, *B. pascuorum* and *B. pratorum* are 'small' species, compared with *B. terrestris* (and *B. lucorum*).

Van der Steen (1994) showed that the acute contact and oral toxicity of dimethoate is correlated with the size of the bumblebee (*B. terrestris*). He investigated the toxicity of

Table III. Sublethal effects of pesticides on honeybees and bumblebees.

Pesticide	Species	Effect	Reference
Diazinon, carbaryl, resmethrin	*A. mellifera*	Decreased longevity and foraging age carbaryl > resmethrin > diazinon, newly emerged workers more sensitive to effects than 14 day workers	Mackenzie and Winston, 1989
Cypermethrin (Fastac)	*A. mellifera*	Repellency for 2 days after treatment on flowering mustard and artificial aphid honeydew on winter wheat	Shires et al., 1984; Shires et al., 1984a
Dicofol	*A. mellifera*	Decreased rate of learning task dependent response	Stone and Willmer, 1989
Deltamethrin (1/27 LD_{50})	*A. mellifera*	Disruption of homing flight, 81% took > 3 times as long as controls to return to nest within time	Vandame, 1995
Parathion (< 0.03 µg/bee)	*A. mellifera*	Disruption of communication dance of foragers on vertical plane due to effects on gravity receptors	Schricker and Stephen, 1970; Stephen and Schricker, 1970
Permethrin (25% LD_{50})	*A. mellifera*	Retards learning (classical conditioning of proboscis extension) for 3 days, beesbees trained prior to exposure no effect	Mamood and Waller, 1990
Permethrin, methoxychlor, malathion, diflubenzuron, carbaryl	*A. mellifera*	Decreased house cleaning (accumulated debris, dead bees)	Nation et al., 1986
Permethrin (0.001 µg/bee)	*A. mellifera*	Significantly increased self-cleaning, trembling dance, abdomen tucking, rotating, and cleaning abdomen, less walking, body insertion, food giving or foraging	Cox and Wilson, 1984
Permethrin	*A. mellifera*	Decreased foraging on treated sweet corn	Pike et al., 1982

Table III. (Continued).

Pesticide	Species	Effect	Reference
Pyrethroids (survivors at LC_{50})	*A. mellifera*	Odor training response affected, max 60% response compared to control flucythrinate = cyfluthrin > permethrin = fenvalerate = cypermethrin > fluvalinate	Taylor et al., 1987
Deltamethrin	*B. terrestris*	At 12.5 g/ha on white mustard in bloom, very low losses bumblebees	Tasei and Carre, 1987
Deltamethrin	*B. terrestris*	0.08–0.16 mg/kg topical application increased sucrose uptake by 40–100%. 0.1–0.2 mg/kg in sucrose decreased uptake by 47–59%, no effect 0.01–0.2 mg/l sucrose on production of workers by queen	Tasei et al., 1994
Range of pesticides (2X highest recommended rate for flowering crops)	*B. terrestris*	No effect pirimicarb, propoxur, endosulfan, phosalone, fenoxycarb, high mortality-parathion, high mortality long term effect-oxydemeton methyl, high mortality repellent effect-deltamethrin, lambdacyhalothrin, moderate mortality-dimethoate	Gretenkord and Drescher, 1993
Permethrin, cypermethrin	*A. mellifera*	Repellency-transitory inhibition of activity (fully reversible in 24 h) following contact, no permanent effects on memory function or foraging efficiency	Reith and Levin, 1988
Deltamethrin	*B. terrestris*	Decreased uptake of treated sucrose by approx. 50%	Tasei et al., 1994

dimethoate to five size classes of bumblebees ranging from 0.162 g to 0.297 g. When corrected for weight, bees in the mid-range (0.168–0.285 g) have similar LD_{50} values 33–37 μg/g bee but small bees (0.162 g) have a lower LD_{50} at 25 μg/g bee and large bees (0.297 g) have a higher LD_{50} at 44 μg/g bee. Therefore although correcting for size can reduce the variability in the mid-range of size, significantly smaller and significantly larger bees have differing LD_{50}s in terms of weight. It is therefore important that data for bumblebees are quoted in terms of weight of the bees tested as, unlike honeybees, their weight can vary significantly between individuals.

Size may also influence oral exposure of bumblebees to pesticides. The quantity of nectar which a bumblebee drinks from a flower has been investigated by Prys-Jones (1982). Uptake rate and total volume of sugar solution imbibed were found to vary according to body weight.

The uptake rate is positively correlated to body size, such that larger bees can drink faster – in general, a doubling of body weight led to a 30–40% increase in uptake rate (Prys-Jones, 1982). The total quantity of nectar drunk also depends on body weight in some species, particularly *B. pascuorum*. Lighter bees can take proportionally larger loads than heavier bees. In real flowers long-tongued bees have been recorded to be faster drinkers than short-tongued bees of similar body size (Harder, 1983).

3.5. Secretion of pesticides into nectar and pollen

There are two possible routes of exposure of bumblebees to pesticides, through uptake of nectar or pollen into which the pesticide has been secreted or through contact with treated foliage or flowers. Systemic pesticides are most likely to occur in nectar and pollen and in nectar their concentrations depends on both the amount and method of secretion. A number of studies have shown contamination of nectar following pesticide exposure (Davis and Shuel, 1988). Investigations with dimethoate and carbofuran in bugleweed (*Ajuga reptans*), oilseed rape (*Brassica napus*) and field beans (*Vicia faba*) showed an apparent selective transport of the insecticides into the nectar as the concentration in nectar often exceeded that in the solution in which the excised flowers were exposed, i.e. it is more than passive movement with water (Davis and Shuel, 1988). There are several reports that it is not only the truly systemic pesticides which can be detected in nectar but also penetrating chemicals such as parathion can result in toxic nectar for up to 24 hours (Jaycox, 1964). Even systemic granular insecticides can penetrate sufficiently into nectar to kill bees although to a far lower extent (Jaycox, 1964).

4. RISK ASSESSMENT

This review suggests that bumblebees are potentially exposed to a wide range of pesticide applications to crops and spray drift on flowering weeds in or near crops. Due to their smaller size, particularly early in the season, bumblebee colonies are more sensitive to impacts on worker numbers than honeybees. Risk assessments for honeybees can be shown to apply to a single bumblebee species for a small number of insecticide classes. A larger database is required on the relative toxicity of fungicides, herbicides and IGRs to bumblebees compared to honeybees to provide more confidence in the extrapolation. These data can be readily generated for *B. terrestris* given their widespread availability due to their commercial use. However, there are few data available on the relative sensitivity of *B. terrestris* compared with other species likely to be exposed and these data need to be generated for a small number of carefully selected, representative, pesticides to ensure protection of these species.

There is also a scarcity of quantitative data on the exposure of bumblebees to pesticides or data which can be readily extrapolated from honeybees. This results in the estimates of exposure being limited to extrapolation from qualitative data or from residue data from other invertebrate species.

Risk assessments in Europe are routinely based on the toxicity-exposure ratio (TER) i.e. the toxicity of the compound (mg/kg) and the dose available (mg/kg), usually by oral intake (EU 91/414). However, for honeybees an empirical approach (hazard quotient) has been developed (EPPO, 1993) based on the application rate of the pesticide (g ai/ha) and the toxicity to the bee (μg/bee). These approaches are reviewed below together with other methods of assessing exposure which may be more readily adapted for use with bumblebees.

4.1. Hazard quotient

A hazard quotient (application rate (g ai/ha)/LD_{50} (μg/bee)) of < 50 is used to define a pesticide as harmless to honeybees, 50–2500 as slight to moderately toxic and > 2500 as dangerous to bees (EPPO, 1993). This approach does not take into account the size/weight of the bee or the route of exposure. For example, an application rate of 15 g ai/ha for alphacypermethrin gives a hazard ratio of 88 for bumblebees and 500 for honeybees, purely due to the difference in toxicity for the individual. Furthermore, this use of a hazard ratio also does not take into account the differences in foraging behaviour and thus exposure of bumblebees.

4.2. Insect residue based data

Pesticide residues on large insects (mg/kg) are calculated by Kenega (1973) as 2.7× application rate in kg/ha and are mainly used for assessing intake in insectivorous birds and mammals (EU 91/414). In this TER approach the residue is compared to the LD_{50}. Therefore, at an application rate for alphacypermethrin of 0.015 kg/ha the residue on a large insect would be 0.041 μg/g. Taking the toxicity of alpha cypermethrin to bumblebees of 0.81 μg/g bee this gives a TER of 20 and a medium risk classification. The classification for honeybees would be TER of 7, a high risk classification. However, this approach is solely based on external residue data and does not take into account the direct uptake on contaminated nectar by bees or the differences in foraging behaviour between species.

4.3. Bee residue data

There have been few quantitative studies to assess the exposure of bees to pesticides. Koch and Weiber (1997) reported the results of a field study using a fluorescent tracer to assess the exposure of honeybees to pesticides applied to crops. The only route which could be assessed in this manner is contact exposure but it provides information which may be extrapolated to other species with similar behaviour patterns.

Using the data produced by (Koch and Weiber, 1997) to produce a dose/toxicity (D/T) for alphacypermethrin at 15 g ai/ha gives a residues of 35 ng/bee (0.175 μg/g bee) and D/T of 0.22 for bumblebees and a residue of 13.5 ng/bee (0.135 μg/g bee) and D/T of 0.45 for honeybees. These are both high risk classifications. This method allows the larger surface area of the bumblebee to be taken into account in the risk assessment. However, further data are required to determine the scale of difference in contact exposure between bumblebees and honeybees based on their behaviour, e.g. number of foraging trips and number of flowers visited. This could be obtained by the same method as (Koch and Weiber, 1997) using a non-toxic dye. There is a need however to develop a model of both contact and oral exposure to pesticides to allow extrapolation between honeybees and bumblebees.

4.4. Pesticide incidents

The submission of bumblebees to the Wildlife Incident Investigation Scheme may provide information on the scale of any problem. Only three incidents involving bumblebees have been reported and pesticides detected, one each in 1995, 1996 and 1997.

1. In 1995 the incident involved dimethoate (0.29 μg/bee dimethoate, 0.12 μg/bee omethoate) and may have been linked to an application to oilseed rape in full flower (misuse of the pesticide) but this could not be confirmed.

2. In 1996 0.033 μg/bee lambda cyhalothrin was detected in dead bumblebees after an application to field beans in full flower (a misuse of the pesticide). Although honeybee colonies were situated nearby no deaths occurred.

3. In 1997 alphacypermethrin (0.0044 μg/bee) was detected in dead bumblebees which had been foraging on oilseed rape which had been sprayed whilst in flower. The spray was applied at 1915 and 1930 (i.e. evening) and contained a mixture of alphacypermethrin, carbendazim and iprodione.

Therefore of the three incidents reported in which pesticides were detected two were apparent misuse (spray application to a flowering crop) and only one followed normal use. The latter demonstrates the potential for exposure of bumblebees at the time when spraying is recommended as no honeybee colonies were affected. It is likely that there are far more bumblebee deaths than the levels reported through the Scheme. These data show that the incident reporting scheme cannot be relied upon to reflect the level of incidents involving bumblebees.

Many of the reported pesticide incidents involving honeybees probably also result in mortality of bumblebees and, together with a reduction in suitable habitat, these have contributed significantly to the decline in bumblebees in the UK over the last 20 years. Applications of sprays to flowering crops or weeds at times when honeybees are less active are likely to result in unreported bumblebee deaths. There is a need to protect foraging bumblebees from direct overspray during the early morning and late evening when pesticides which are repellent but highly toxic are applied, i.e. pyrethroids. Of particular concern are those pesticides applied when queens are emerging and establishing colonies, e.g. March/April, when colonies may be significantly impacted by the loss of a small number of workers or the queen. This is a problem which cannot readily be addressed by risk management measures due to differing foraging profiles of honeybees and bumblebees but does need to be taken into account in risk assessment and the development of more selective compounds.

5. CONCLUSIONS

The reliable extrapolation of risk for pesticides from honeybees to bumblebees depends on the extent of variations in exposure and toxicity. Bumblebees use the majority of insect-pollinated crop plants to a large extent and are likely to be exposed to the same pesticide applications as honeybees. In addition, significantly greater exposure of bumblebees is likely to occur due to spray drift onto forage plants in and around field margins when non-flowering crops are sprayed. Of particular concern in extrapolation of risk assessments which are based on applications of sprays, e.g. pyrethroids, to flowering crops or weeds at times when honeybees are less active but bumblebees are active. In conclusion it is likely that exposure of bumblebees is at least that of honeybees and probably greater.

Generally, the toxicity of the pesticides for which data are available are lower to bumblebees than honeybees when expressed on a weight basis. However, it should also be remembered that the data available are limited in terms of number and type of insecticides. There is a need to increase the amount

of toxicity data available for bumblebees in order to support the assumption that they are less susceptible than honeybees.

The behaviour of the species affects also the risk posed. Of particular concern are those pesticides applied when queens are emerging and establishing colonies, e.g. March/April, when colonies may be significantly impacted by the loss of a small number of workers or the queen. This is a concern specific for bumblebees as only the queen and not the entire colony overwinters. These significant differences in behaviour of species cannot be readily addressed by simple risk management methods, e.g. changing timings of applications. Differences in foraging behaviour, ecology etc. need to be taken into account in risk assessment and the development of more selective compounds.

Résumé – Estimation de l'exposition des bourdons (*Bombus* sp.) aux pesticides et de leur toxicité. Il y a eu un déclin important des populations de bourdons dans les 30 dernières années, particulièrement dans le sud de la Grande Bretagne, qui peut être dû en partie à l'utilisation de certains pesticides. Les bourdons sont d'importants pollinisateurs de certaines cultures et de nombreuses plantes sauvages et sont considérés comme des insectes « auxiliaires » (Corbet et al., 1001 ; Williams, 1997 ; Prys-Jones and Corbet, 1991). De nombreux pesticides utilisés actuellement, tels que les pyréthrinoïdes, connus pour être plus toxiques à basse température, sont appliqués tôt le matin ou tard le soir. Ils peuvent donc constituer un grand danger pour les bourdons qui butinent à des températures inférieures à celles auxquelles butinent les abeilles domestiques (Fig. 1). Des notifications de mortalités de bourdons sont peu probables car ces insectes ne sont pas élevés et, leurs colonies étant relativement petites, ils vont mourir en petits nombres. Cette mise au point vise à évaluer les données disponibles concernant l'exposition des bourdons aux pesticides et la toxicité de ces derniers pour les bourdons et d'identifier les approches possibles pour estimer les risques.

Cette mise au point suggère que les bourdons sont potentiellement exposés à un large éventail de traitements pesticides sur les cultures et à des dérives de pulvérisation sur les adventices en fleurs dans ou près des cultures. En raison de leur taille restreinte, en particulier en début de saison, les colonies de bourdons sont plus sensibles que les colonies d'abeilles domestiques à la réduction du nombre d'ouvrières. On a montré que la toxicité des pesticides pour les abeilles domestiques étaient la même que pour une espèce de bourdon donnée (Tab. I) pour un petit nombre de classes de pesticides (Tab. II) avec un large éventail d'effets sublétaux observés (Tab. II, Fig. 2). Il est nécessaire d'établir une large base de données concernant la toxicité relative des fongicides, herbicides et régulateurs de croissance pour les bourdons comparée à la toxicité pour les abeilles, afin de fournir des extrapolations plus fiables. Ces données peuvent être facilement générées pour *B. terrestris* car elles sont largement disponibles en raison de l'utilisation commerciale de ce bourdon. Il y a pourtant peu de données existantes sur la sensibilité relative de *B. terrestris* comparée à celle des autres espèces susceptibles d'être exposées et il est nécessaire de générer ces données pour un petit nombre de pesticides représentatifs et soigneusement sélectionnés afin d'assurer la protection de ces espèces.

Nombre des incidents de pesticides signalés à propos des abeilles domestiques ont probablement entraîné des mortalités de bourdons et causé, avec la diminution des habitats appropriés, le déclin des populations de bourdons dans le Royaume-Uni au cours des 20 dernières années. Les pulvérisations sur les cultures ou les adventices en fleurs, à des périodes où les abeilles domestiques sont moins actives, sont susceptibles de causer des mortalités de bourdons non signalées. Le besoin existe de protéger les bourdons butineurs de la pulvérisation directe

tôt le matin et tard le soir, lorsque sont faits les traitements de pesticides répulsifs mais hautement toxiques, i.e. les pyréthrinoïdes. Une attention particulière doit être portée à ces pesticides appliqués lorsque les reines émergent et fondent les colonies, c'est-à-dire en mars/avril, et que les colonies peuvent être touchées de façon significative par la perte de la reine ou d'un petit nombre d'ouvrières. C'est un problème qui ne peut pas être facilement résolu par des mesures de gestion des risques en raison des profils de butinage différentes des abeilles et des boudons, mais qu'il faut prendre en compte dans l'évaluation des risques et la mise au point de composés plus sélectifs.

***Apis mellifera* / *Bombus terrestris* / plante transgénique / *Bacillus thuringiensis* / inhibiteur de protéase**

Zusammenfassung – Einschätzung der Kontamination und der Giftigkeit von Pestiziden bei Hummeln (*Bombus* sp.). In den letzten 30 Jahren ist das Vorkommen von Hummeln besonders in Südengland drastisch zurückgegangen. Möglicherweise liegt dies zum Teil an bestimmten Pestiziden. Hummeln sind wichtige Bestäuber von einigen Nutzpflanzen und vielen Wildpflanzen und gehören somit zu den Nutzinsekten (Corbet et al., 1991; Williams, 1997; Prys-Jones and Corbet, 1991). Viele zur Zeit benutzte Pestizide wie Pyrethroide sind dafür bekannt, dass sie bei niedrigen Temperaturen giftiger sind und werden daher am frühen Morgen oder späten Abend ausgebracht. Deshalb könnten diese Pestizide eine größere Gefahr für sammelnde Hummeln als für Honigbienen sein, weil diese bei niedrigeren Temperaturen ausfliegen (Abb. 1). Es ist unwahrscheinlich, dass das Sterben von Hummeln bemerkt wird, denn diese werden nicht als Haustiere gehalten und sie sterben nur in geringen Anzahlen, da ihre Völker relativ klein sind. Diese Übersicht versucht eine Auswertung der bekannten Daten über die Kontaminierung von Hummeln mit Pestiziden und ihre Giftigkeit durchzuführen. Außerdem soll eine mögliche Risikoabschätzung vorgenommen werden.

Es wird angenommen, dass Hummeln potentiell einer großen Zahl von Pestiziden ausgesetzt sind, die zur Schädlingsbekämpfung von Feldfrüchten und Obstplantagen eingesetzt werden und zusätzlich auf blühende Wildkräuter und andere Nutzpflanzen in der Nähe verwehen. Wegen ihrer geringen Individuenzahl sind Hummelvölker vor allem im Frühjahr viel empfindlicher für den Verlust einiger Arbeiterinnen als Honigbienen. Es konnte gezeigt werden, dass die Giftigkeit der Pestizide bei Hummeln ähnlich wie bei den Honigbienen ist (Tab. I). Das gilt für eine kleine Zahl von Pestizidklassen (Tab. II) und für eine große Zahl der beobachteten sublethalen Wirkungen (Tab. III, Abb. 2). Eine größere Datenbasis über die relative Giftigkeit von Fungiziden, Herbiziden und IGRs für Hummeln im Vergleich zu Honigbienen ist notwendig, um mehr Sicherheit bei der Extrapolation ihrer Wirkungen zu erhalten. Diese Daten könnten leicht bei *B. terrestris* erhoben werden, da diese durch ihre kommerzielle Nutzung weitverbreitet erhältlich sind. Im Vergleich zu anderen Arten, die wahrscheinlich auch Pestiziden ausgesetzt sind, gibt es inzwischen einige Daten über ihre relative Empfindlichkeit. Diese Daten müssen für eine bestimmte Zahl sorgfältig ausgesuchter und repräsentativer Pestizide erhoben werden, um einen Schutz dieser Arten zu gewährleisten.

Viele der bekannten Schadensfälle bei Honigbienen durch Pestizide haben wahrscheinlich auch zum Sterben von Hummeln geführt, und haben zusammen mit der Abnahme von geeigneten Lebensräumen zur Abnahme der Hummeln in UK in den letzten 20 Jahren geführt. Die Anwendung von Sprühmitteln in blühenden Äcker und Plantagen zu Zeiten, in denen Honigbienen weniger aktiv sind, haben wahrscheinlich unbemerkt zum Hummelsterben beigetragen. Es besteht die dringende Notwendigkeit,

sammelnde Hummeln vor einer direkten Besprühung in den frühen Morgen- und den späten Abendstunden zu schützen. In dieser Zeit werden solche Pestizide ausgebracht, die repellent Wirkung haben aber hoch toxisch sind, z.B. Pyrethroide. Besonders schlimm sind die Pestizide, die in der Phase der Nestgründung durch Königinnen versprüht werden, im März und April. In dieser Zeit sind die Völker besonders durch Verlust von wenigen Arbeiterinnen und der Königin gefährdet. Dieses Problem kann nicht so leicht durch Risikomanagement gelöst werden, da das Sammelverhalten von Honigbienen und Hummeln unterschiedlich ist. Aber es muss dringend bei der Risikoabschätzung und bei der Entwicklung von noch spezifischeren Verbindungen berücksichtigt werden.

Hummeln / Pestizide / Kontamination / Risikoabschätzung

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