Critical Perspectives

The Impact of Pesticides on Flower-Visiting Insects: A Review with Regard to European Risk Assessment

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Abstract: Flower-visiting insects (FVIs) are an ecologically diverse group of mobile, flying species that should be protected from pesticide effects according to European policy. However, there is an ongoing decline of FVI species, partly caused by agricultural pesticide applications. Therefore, the risk assessment framework needs to be improved. We synthesized the peer-reviewed literature on FVI groups and their ecology, habitat, exposure to pesticides, and subsequent effects. The results show that FVIs are far more diverse than previously thought. Their habitat, the entire agricultural landscape, is potentially contaminated with pesticides through multiple pathways. Pesticide exposure of FVIs at environmentally realistic levels can cause population-relevant adverse effects. This knowledge was used to critically evaluate the European regulatory framework of exposure and effect assessment. The current risk assessment should be amended to incorporate specific ecological properties of FVIs, that is, traits. We present data-driven tools to improve future risk assessments by making use of trait information. There are major knowledge gaps concerning the general investigation of groups other than bees, the collection of comprehensive data on FVI groups and their ecology, linking habitat to FVI exposure, and study of previously neglected complex population effects. This is necessary to improve our understanding of FVIs and facilitate the development of a more protective FVI risk assessment. Environ Toxicol Chem 2019;00:1–16. © 2019 The Authors. Environmental Toxicology and Chemistry published by Wiley Periodicals, Inc. on behalf of SETAC.

Keywords: Pollinator insects; Bees; Exposure; Effects; Regulatory deficits; Regulatory development

INTRODUCTION

The evidence that flower-visiting insects (FVIs) are in decline is continuously growing (Goulson et al. 2015; Potts et al. 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Hallmann et al. 2017; Ollerton 2017; Powney et al. 2019; Vray et al. 2019). This is apparent in losses of domestic honey bee (Apis mellifera) hives in the many European Union countries and the United States of America along side a simultaneous decline in wild bee diversity and butterfly, moth, and syrphid fly populations (vanEngelsdorp et al. 2008; Goulson et al. 2015; Potts et al. 2015; Powney et al. 2019; Vray et al. 2019). Hallmann et al. (2017) showed a substantial long-term decline in flying insect biomass in nature reserves, which included many flower visitors such as butterflies, bees, flies, and beetles. This general decrease in species and abundances is caused by multiple, mostly anthropogenic, factors, one of which is exposure to pesticides (Goulson et al. 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Ollerton 2017). Other causes discussed include habitat loss and fragmentation, resource diversity decrease, climate change, parasites and pathogens, invasive species, and environmental pollution (Goulson et al. 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Ollerton 2017).

The FVIs provide a vital ecosystem service (pollination) that propels human food production and maintains flowering plant biodiversity (Klein et al. 2007; Ollerton et al. 2011). However, FVI protection is relevant not only for the protection of commercial yield and native flora, but also because FVIs form a major part of faunal biodiversity, approximately 30% of all arthropod species worldwide (Wardhaugh 2015).

According to the Convention on Biological Diversity (United Nations 1992), biodiversity should be protected. In 2016, the United Nations specifically called for pollinator conservation in agriculture in their Cancun Declaration (United Nations 2016). This resulted in the formation of the continuously growing Coalition of the Willing on Pollinators that commits to...
protecting pollinators and their habitat from harmful anthropogenic impact (Coalition of the Willing on Pollinators 2016). In Europe, Regulation (EC) 1107/2009 is in place, concerning the regulatory risk assessment framework to prevent unacceptable negative impacts of agricultural pesticide use on biodiversity (European Commission 2009). Therefore, FVI protection from significant adverse pesticide effects is required by European law. Because pesticides contribute to the ongoing FVI decline, it is possible that regulatory measures are insufficient to provide protection from pesticides (Goulson et al. 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Ollerton 2017).

For the present review, we summarized the available scientific literature and regulatory documents to examine the impact of pesticides on FVIs. We further discuss the suitability of European risk assessment to prevent adverse consequences of pesticide use. Species decline has mostly been noted in bees because of an economic interest in preserving viable populations of these important pollinators (Klein et al. 2007; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016). However, many other FVI taxa are exposed to pesticides in the agricultural landscape, which may lead to negative effects on their populations (Godfray et al. 2014, 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Potts et al. 2015). Consequently, we identified the relevant FVI groups by visitation frequency and abundance in crops and the surrounding area and describe their ecology and habitat. We used this knowledge to characterize preferential pesticide exposure pathways and to summarize quantitative exposure of relevant habitat compartments from residue studies. We collated effects studies to assess the impact of environmentally realistic pesticide doses on FVIs. This enabled us to critically discuss the suitability of the regulatory effect and exposure assessment. We further propose data-driven tools that improve FVI risk assessment. Finally, we show the major knowledge gaps that need to be closed to increase our understanding of FVIs and develop a sufficiently protective regulatory framework.

**REVIEW METHODOLOGY**

We searched the peer-reviewed English-language literature published until 2018 using Google Scholar. Keywords included the following terms and their combinations: “pollinator,” “flower visiting insect,” “bee,” “butterfly,” “moth,” “fly,” “beetle,” “habitat,” “trait,” “pesticide,” “insecticide,” “risk assessment,” “exposure,” “residue,” “effect,” and “toxicity.” We also considered Researchgate (2016) suggestions and results of a continuous Sparrho (2013) search that used the keywords “pollinator,” “pesticide,” and “bee.” Other studies were brought to our attention through recommendations from scientific colleagues, and we obtained additional papers from literature references. Semi-field and field effect studies were only included if they investigated the impact of environmentally realistic pesticide exposure levels. Finally, we screened European Union regulatory documents to gain detailed knowledge about European risk assessment for bees and non-target arthropods (NTAs).

**FVI GROUPS**

In the context of the present review, FVIs are defined as insect species that directly interact with flowers in at least the flying adult life stage, in accordance with Wardhaugh (2015). Most so-called pollinators have actually only been determined to be FVIs because the usual visual observations on flowers are not suitable to prove pollen deposition on the stigma, that is, pollination. The FVIs are an ecologically complex aggregation, which includes species with very different life strategies, for example, herbivores, predators, and parasites (Ollerton 2017). To assess the impact of pesticides on FVIs, it is important to identify the relevant groups that frequently visit flowers and are abundant in the agricultural landscape. In the past, the scientific literature sparsely identified FVI groups aside from bees. Lepidopterans (moths and butterflies), and flies (mainly hover flies) are acknowledged as important taxa. Beetles and wasps are mentioned as flower visitors of minor importance (Winfree et al. 2011). In recent years it was hypothesized that there are considerably more FVI species than previously assumed. The probable FVI groups span from bees, moths and butterflies, beetles, wasps, and ants to flies but also include less prominent groups such as thrips, true bugs, springtails, termites, and cockroaches (Wardhaugh 2015; Ollerton 2017). However, these classifications were only based on estimates and needed support from field research.

Several studies found that FVI communities in the agricultural landscape are indeed as diverse as theoretically suggested. Wildflower plantings were visited by many insect taxa aside from bees, lepidopterans, and hover flies in the central German agricultural landscape (Grass et al. 2016). In fact, non-bee/non-hover fly insects made up half of the visiting individual visits and 75% of FVI species (Figure 1). Non-hover fly dipterans were by far the largest portion of visiting species. In contrast, butterflies only made up a small share of FVI abundance, whereas the numbers of flower visits by beetle and non-hover fly dipteran individuals was comparable with those of honey bees (Grass et al. 2016). A large-scale meta-analysis also found that non-hover fly dipterans are at

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FIGURE 1: Wildflower planting flower visitations in central Germany. The dashed line shows the cumulative fraction of honey bee and hover fly flower visits. Adapted from Grass et al. 2016.
least as relevant as hover flies because they made up the majority of dipteran flower visits (Orford et al. 2015). Such a distribution was also found in a common plant of the agricultural landscape that is widely distributed in Europe. The common buttercup *Ranunculus acris* was mostly visited by fly species, as shown in a large-scale, long-term project, Biodiversity Exploratories (Fischer et al. 2010; Figure 2). Beetles were detected in similar species and individual numbers as bees (Fischer et al. 2010; Nico Blüthgen, Technical University Darmstadt, Darmstadt, Germany, personal communication, 2012).

The diversity of FVI communities has been underestimated not only for native flora but also for crops. An extensive meta-analysis summarized the results of 39 field studies that investigated flower visits in several crop systems from 5 continents (Rader et al. 2015). Overall, non-bee species accounted for 38% of flower visits. The visits by non-bees of oilseed rape as a typical European mass-flowering crop were quite variable (5–80%) and varied even within countries (5–60%).

The FVI communities are far more diverse than it has been acknowledged in the past. In general, visit rates vary greatly between cropping systems, native habitats, and geographic locations (Orford et al. 2015; Rader et al. 2015; Grass et al. 2016). The available literature identifies the relevant European FVI groups in crops and their semi-natural surroundings as bees, flies (non-syrphids and syrphids), lepidopterans (moths and butterflies), and beetles. However, only limited information is available to evaluate all groups. Therefore, it is currently not possible to assess the relevance of all other suspected groups, for example, non-bee hymenopterans and hemipterans. After the identification of relevant FVI groups, it is necessary to examine their ecology and their habitat to assess their potential pesticide exposure.

**FVI ECOLOGY, HABITAT, AND EXPOSURE PATHWAYS**

**Ecology**

Other than visiting flowers in at least their adult stage, FVIs differ substantially in their ecology (Ollerton 2017). A comprehensive review of FVI ecology is beyond the scope of the present review. We therefore concentrate on bee species because they are extensively studied in the ecotoxicological context and cover many of the general FVI traits. Specific additional properties of other groups will also be mentioned.

All bee species are obligate florivores in larval and adult life stages. This distinguishes them from all other FVI taxa for which only a subset of species are flower visitors and mostly adults are florivores. Adult bees feed predominantly on nectar, whereas larvae feed mostly on pollen (Michener 2007). Other FVI groups such as moths and butterflies and beetles also have herbivore life stages (Koch and Freude 1992; Ebert 1994; Scoble 1995). Aside from the well-known domesticated western honey bee *Apis mellifera*, there are a multitude of ecologically variable wild bee species in Europe. Some species are eusocial, that is, they live in colonies or aggregations, but most species are solitary. In addition, there are many parasitic species that exploit their bee host to feed and tend to their offspring (Westrich 1990; Michener 2007). Bees have several nesting strategies. Most species burrow into the soil to build their nests, but others also excavate deadwood, occupy pre-existing cavities in soil or deadwood, or they construct nests from collected material (Michener 2007). Other FVI groups also contain soil-dwelling larval stages, for example, flies and beetles (Koch and Freude 1992; Frouz 1999). There are food generalists (polylectic) and specialists (oligolectic) that in some cases forage on just one specific plant (Westrich 1990; Michener 2007). The active flight period and length of flight differs between bee species. Many species start mating and foraging flights in spring whereas others do not begin their adult phase before summer and continue until autumn (Westrich 1990). Most species have only one brood/year (univoltine), whereas some lay eggs throughout the year (multivoltine). Voltinism varies with geography and climate (Michener 2007). Daily activity usually peaks at mid-day but can also peak in the morning and evening (Thompson 2001; Steen 2016). Bee species vary greatly in their foraging range, that is, the distance they can cover to search for food resources, which ranges from hundreds of meters to 10 or more kilometers (Zurbuchen et al. 2010).

**Habitat**

Due to their ecological profile, FVI species need a set of compartments inside a habitat to fulfill basic needs: food, water, shelter, mating space, and nesting grounds (Table 1). The agricultural landscape generally comprises viable habitats that can be categorized as crop plantings and nontarget areas, for example, managed flower strips and field edge structures (Marshall and Moonen 2002; Hahn et al. 2015; Tschumi et al. 2015). These areas differ in many aspects such as structure, plant species inventory, spatial and temporal food resource availability, natural enemies, or anthropogenic stress (Marshall and Moonen 2002; Hahn et al. 2015; Tschumi et al. 2015). Therefore, habitat quality varies significantly, which theoretically enables us to assess habitat attractiveness for FVIs.

Numerous crops have been classified as bee attractive (Supplemental Data, Table S1; European Food Safety Authority 2013). However, it is currently not possible to quantitatively evaluate the suitability of a certain crop as a FVI food source. Most studies were
only performed with honey bees and focus on major sources of pollen/nectar in their diet rather than the food spectrum (European Food Safety Authority 2013). Mass-flowering crops such as oilseed rape Brassica napus and sunflower Helianthus annuus are used as food sources by wild and managed bees. Their overabundant supply of floral resources will be used to some degree even if they are not the preferred food plant of an FVI species (Holzschuh et al. 2013; Coudrain et al. 2015; Requier et al. 2015). Even virtually non-attractive crops plantings such as corn or cabbage might be FVI habitats if there is undergrowth of crop-associated wild plants, for example, cornflower or poppy species (Storkey and Westbury 2007; Balmer et al. 2014; Manandhar and Wright 2016). Furthermore, crops can still provide habitat functions for FVIs even if they are not flowering, for example, as nesting grounds or temporary refuge.

Aside from crops, there are non-target areas that are used as habitat by FVIs. Field edge structures are semi-natural habitats in intensely managed agricultural areas. They provide multiple habitat functions for FVI species such as refugia, feeding and breeding grounds, and migration corridors for FVI species (Marshall and Moonen 2002; Marshall et al. 2006; Denisow and Wrzesień 2015). Flower strips are sown with seed mixtures for insect conservation, with an emphasis on sustaining pollinator populations. They ensure crop pollination and also favor predacious beneficials to support biological pest control (Haaland et al. 2011; Feltham et al. 2015; Tschumi et al. 2015). These nontarget areas, however, have also not been adequately studied to discuss their habitat suitability in more detail.

In the absence of sufficient information and to exercise the precautionary principle, we assume in the present review that the entire agricultural landscape is FVI habitat. Therefore, FVIs may potentially be exposed to pesticides while interacting with habitat compartments of crop and non-target areas.

### Exposure pathways

Pesticides are transported into FVI habitats by direct application to crops (primary processes) or unintentional redirection of a fraction of the applied pesticide amount into adjacent areas (secondary processes; Figure 3). Primary processes include spray and solid application, by seed treatment or granules, for example (Walker 2001; Nuyttens et al. 2013). Stem application and irrigation methods play a minor role in Europe (Düker and Kubiak 2015; Miorini et al. 2017). Secondary processes are spray drift, field-edge overspray, dust dispersion, and run-off. As a result of this pesticide input into crops and non-target areas, all FVI habitat compartments are potentially contaminated (Sgolastra et al. 2019).

Exposure of airspace, pollen and nectar, stems/leaves, soil, and water sources (rivers/lakes, puddles, guttation water) can subsequently lead to FVI exposure (Figure 3). Pesticide applications on less attractive crops can still cause FVI exposure if there is flowering weed undergrowth (e.g., cornflower or poppy species in cereal fields), or by transport into attractive off-crop areas (Botías et al. 2015; Simon-Delso et al. 2017). The identification of potentially contaminated habitat compartments does not allow for an estimation of FVI pesticide exposure. It is therefore necessary to quantify the exposure of habitat compartments and link it to FVI contamination to identify important pathways.

### EXPOSURE TO PESTICIDES

#### Individuals

Investigations of pesticide residues levels in FVI individuals are required to assess pesticide exposure. Unfortunately, these data are only available for bees at the moment. Most bee exposure studies in recent years have investigated the chemical

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**TABLE 1**: Habitat compartments used by flower-visiting insects

| Compartment        | Life stage                          | Function                                                                 |
|--------------------|-------------------------------------|-------------------------------------------------------------------------|
| Airspace           | Adults                              | Food search (foraging), mate search, nest search                       |
| Flowers            | Adults and florevore larvae         | Food collection (foraging), shelter, mating, nesting, nest material collection |
| Stems/leaves       | Adults and herbivore larvae         | Nesting, shelter                                                        |
| Soil               | Adults and soil-dwelling larvae     | Water collection/consumption                                            |

**FIGURE 3**: Exposure pathways from application to habitat compartments in in- and off-crop habitats. Yellow up/down arrows indicate primary and pink side/down arrows secondary transport processes.
class of neonicotinoids. Furthermore, the vast majority of these studies is concerned with honey bee exposure (Blacquière et al. 2012; Godfray et al. 2014, 2015; Bonmatin et al. 2015; Wood and Goulson 2017). Hence, such research is overrepresented compared with other pesticide classes or bee species in the following sections.

Bees are exposed to a plethora of pesticides. Brood and adult bee samples from North American honey bee colonies contained 46 pesticides of different pesticide classes and their metabolites (Mullin et al. 2010). A French study found residues of 19 compounds in honey bee colony samples (Chauzat et al. 2011). All major pesticide classes are detected in honey bees (insecticides, fungicides, and herbicides), according to the comprehensive list compiled by the EFSA Authority Plant Protection Products and Their Residues Panel (2012). A more recent study investigated pesticide residues in differentbumble bee species and found at least one insecticide or fungicide in over half of the analyzed individuals (Botías et al. 2017). The majority of these individuals were exposed to multiple compounds.

**Nectar and pollen**

Nectar and pollen are major carriers of pesticide loads for FVIs. The North American and French studies mentioned in the previous Individual sections found residues of 98 and 19 pesticides and metabolites in collected pollen, respectively (Mullin et al. 2010; Chauzat et al. 2011). A more recent Italian study registered 18 different insecticides and fungicides in pollen over a 3-yr sampling period (Tosi et al. 2018). The maximum neonicotinoid residues in pollen and nectar were determined to be $10^6$ to $10^7$ ng/g and $10^3$ ng/g, respectively (Goulson 2013; Godfray et al. 2014; Wood and Goulson 2017). Residue levels fluctuate between crops by one order of magnitude, but pollen doses are consistently higher than nectar doses (Wood and Goulson 2017; Gierer et al. 2019). Several parameters such as dose and mode of treatment, physicochemical properties of the pesticide, crop type, season, location, soil type, weather, and sampling time of day influence pesticide doses in both matrices (Wood and Goulson 2017; Gierer et al. 2019).

Pesticide load in bee-collected pollen and nectar is often similar to residues in crops (Rundlöf et al. 2015; Wood and Goulson 2017). However, there are also studies that found much lower contamination (Cutler and Scott-Dupree 2014; Rolke et al. 2016). Because bees collect pollen and nectar from a wide variety of plants, the dietary spectrum partly determines their contamination. The highest levels of residues are found when a large proportion of crop pollen is collected (Pohorecka et al. 2013; Cutler and Scott-Dupree 2014; Botías et al. 2015; Rundlöf et al. 2015; David et al. 2016). Noncultivated plants adjacent to crops are often also contaminated with pesticides in greatly variable doses that can reach comparable levels (Botías et al. 2015; Mogren and Lundgren 2016; Wood and Goulson 2017). In general, high doses in nectar and pollen temporally coincide with the bloom of mass-flowering crops such as oilseed rape (Wood and Goulson 2017). However, chronic exposure of species with a long active flight period, such as honey bees or bumble bees, might be driven by wildflower foraging. One study found that 97% of total neonicotinoid residues in pollen in June and August were actually derived from wildflowers (Botías et al. 2015).

**Soil**

The majority of European bee species (60–70%) nest in soil either by actively burrowing nests or using existing cavities (Westrich 1990). Therefore, pesticide exposure by soil contact may be an important, yet underestimated pathway (Gradish et al. 2019; Sgolastra et al. 2019). Soil exposure may also be relevant for soil-dwelling life stages of other FVI groups such as fly and beetle larvae (Koch and Freude 1992; Frouz 1999). Systemic pesticides are usually applied directly to the soil to be taken up by crops. Only a fraction of the applied pesticide load enters the plant body whereas the major part remains in the soil (Sur and Stork 2003; Alford and Krupe 2017). Agricultural soils are therefore often contaminated with multiple pesticides (Hvězdová et al. 2018). Measurable neonicotinoid residues in various crop soils range from $10^{-1}$ to $10^1$ ng/g (Jones et al. 2014; Botías et al. 2015; Heimbach et al. 2016; Wood and Goulson 2017). To assess pesticide exposure, it is important to know not only the (peak) concentrations but also the persistence in the soil matrix. Half-lives of neonicotinoid insecticides range from several days to years (Goulson 2013). Values greater than 1 yr suggest possible accumulation or continuing exposure from applications in previous years. Both scenarios have been demonstrated for neonicotinoids by chemical analysis of crop soils (Bonmatin et al. 2005; Goulson 2013; Jones et al. 2014).

**Stem/leaves**

Systemic pesticides are designed to be taken up by crops from the soil. Depending on the crop, 1.6 to 20% of the applied amount of neonicotinoids is absorbed into the plant body (Alford and Krupe 2017; Sur and Stork 2003). Several studies have also found neonicotinoid residues in wild plant stems or leaves from field margins at levels of $10^0$ to $10^2$ ng/g (Pecenka and Lundgren 2015; Botías et al. 2016; Mogren et al. 2016). Exposure of FVIs by stem or leaf material may not be restricted to herbivore life stages. Because they use the plant body as a refuge or collect parts of it as nesting material (as do leaf cutter bees; *Megachile* spp.), FVI adults might also be exposed to pesticide residues by contact (Sgolastra et al. 2019).

**Water sources**

The FVIs can potentially take up pesticides from different water sources. Ephemeral puddles on farmland have been shown to contain maximum neonicotinoid concentrations of $10^1$ ng/mL that may represent a risk to bees (Samson-Robert et al. 2014; Schaafsma et al. 2015). Another potential water source for FVIs are guttation droplets that are exuded by some plant species under moist conditions. Concentrations of systemic neonicotinoids in crop guttation fluid vary greatly
Tapparo et al. 2011; Reetz et al. 2016; Wirtz et al. 2018). Maximum concentration have been measured at 10^5 ng/mL (Godfray et al. 2014; Schmolke et al. 2018). Exposure at toxicologically relevant doses is only expected in crops treated with systemic pesticides, because spray treatments lead to doses that are lower by 3 orders of magnitude (Bonmatin et al. 2015). In addition, there is first evidence that seed treatment of crops can lead to contamination of guttation fluid in weeds that grow in proximity (Mörtl et al. 2019). Field-adjacent rivers and lakes are heavily contaminated with pesticides at levels that often present a risk for aquatic invertebrates (Morrissey et al. 2015; Stehle and Schulz 2015). Exposure through surface waters might also be toxicologically relevant for bee species (Sánchez-Bayo et al. 2016).

**Linking habitat to individual exposure**

The FVI habitats in crops and nontarget areas are exposed to pesticides. However, it is generally difficult to connect the exposure of these habitats to the contamination of FVI individuals. For nectar, pollen, and stem/leaf material, this would require that FVI food intake be broken down and quantified. Bee adults usually procure their energy from carbohydrate-rich nectar, whereas larvae feed on pollen provision/pollen bread, a mixture of mostly protein-rich pollen and minor nectar content (Westrich 1990). Because polylectic bee species forage on a wide variety of plant species (Coudrain et al. 2015; Sickel et al. 2015), their larval pesticide uptake is highly dependent on the proportion of contaminated nectar and pollen in their diet. Data on the FVI food spectrum and corresponding pesticide exposure are scarce. There are some quantitative estimates of adult and larval bee food consumption, but it is not clear how this would translate into an individual bee pesticide load (European Food Safety Authority 2013). Food intake varies greatly between bee species, which makes it impossible to generalize single-species estimates (Müller et al. 2006). In addition, there is insufficient information to connect stem or leaf exposure to FVI contamination (Sgolastra et al. 2019).

Linking soil to FVI exposure is even more difficult. Pesticides can be sorbed to the soil and become bound residues with decreased bioavailability and degradation rates, especially when the chemicals are hydrophobic (Gevao et al. 2000; Semple et al. 2003). Water-soluble compounds such as neonicotinoids might not be so prone to sorption and might therefore retain their bioavailability to a greater extent. There is currently no approach toward estimating FVI exposure after soil contact.

The details of FVI water uptake are nearly unknown. There are estimates of the daily water intake of the honey bee and one wasp species (European Food Safety Authority 2013). Still, the majority of FVI species have not been studied, and it is unclear which water sources are used and to what degree. In the case of guttation, it has been stated that this phenomenon rarely occurs in most crops, especially in high enough concentrations to be of toxicological relevance (Schmolke et al. 2018; Wirtz et al. 2018). This may also be true for exposure via surface waters and puddles. There is currently no clear link between pesticide residues in the available water sources and pesticide uptake of FVIs (Wood and Goulson 2017).

Because FVIs are exposed to pesticides in their habitat, the subsequent effects need to be assessed to evaluate the consequences for FVI populations and communities.

**PESTICIDE EFFECTS**

**General considerations**

To determine the risk of pesticide applications for FVIs, it is necessary to investigate their sensitivity to such chemicals. Only detailed information for a representative amount of species would allow an assessment of the entire group of FVIs. Because the honey bee is a test organism in European pesticide risk assessment, there are extensive acute toxicity data on all registered pesticides for this species. However, other bee species’ sensitivity to pesticides is practically unknown and may differ substantially.

The European Commission restricted the use of the neonicotinoids imidacloprid, clothianidin, and thiamethoxam in 2013 because of high acute risks for bees. Since then, several complex semi-field and field studies have been carried out to investigate neonicotinoid effects on honey bees, non-Apis, and wild bee species at environmentally realistic exposure levels. Unfortunately, there is still nearly no information on pesticide effects on all other non-bee FVI groups. Several colony-level honey bee studies found limited to negligible effects after neonicotinoid exposure (Pilling et al. 2013; Cutler et al. 2014; Dively et al. 2015; Rundlöf et al. 2015). Honey bee effects from these studies are hardly translatable to all other European bee species because of substantial ecological differences, mainly social structure and sheer individual numbers in a population (Stoner 2016; Wood and Goulson 2017). Therefore, honey bee field effects will not be elaborated on in the present review. See the following review articles for further information on honey bee field effects: Blacquière et al. (2012), Godfray et al. (2014, 2015), Goulson (2013), and Pisa et al. (2017, 2015).

**Semi-field studies**

**Reproduction.** Several (semi)-field studies have investigated non-Apis bee reproduction and colony growth effects in similar experimental setups, mostly with Bombus terrestris. Bumble bee colonies were exposed either by feeding them contaminated nectar but letting them forage without restriction or setting them up next to farmland to which pesticides had been applied. To summarize, in most studies neonicotinoid exposure led to reductions in worker, male, and queen offspring (colony growth), reduced individual growth, and skewed sex ratio (Gels et al. 2002; Whitehorn et al. 2012; Cutler and Scott-Dupree 2014; Moffat et al. 2015, 2016; Rundlöf et al. 2015; Ellis et al. 2017; Main et al. 2018). Impaired reproduction is caused not only by neonicotinoids but also by application of new substance classes such as sulfoximine insecticides (Siviter et al. 2018). Only one study examined field effects on solitary bees and recorded a total reduction in brood cell construction by
Osmia bicornis next to clothianidin-treated oilseed rape (Rundlöf et al. 2015). However, there are also a few studies that found no adverse effects on bumble bees and solitary bees in field settings (Peters et al. 2016; Sterk et al. 2016; Ruddle et al. 2018). Discrepant outcomes between these and the majority of studies most likely result from different exposure levels. In comparison to Rundlöf et al. (2015), these 3 studies used a very similar setup: bumble bee colonies and solitary bee trap nests were placed next to seed-treated oilseed rape. However, Sterk et al. (2016) and Peters et al. (2016) used the winter variety of oilseed rape and Rundlöf et al. (2015) used the spring variety. This resulted in a nearly 10-fold difference in maximum pollen residues, which is a highly likely cause for the contrasting effects.

Foraging. A general pattern has emerged showing that the number of bee trips to flowers increases but foraging efficiency decreases after pesticide exposure. Pesticide effects on bumble bee foraging were investigated in (semi-)field studies similar to the experiments just described in the Reproduction section. However, bees were exposed through pesticide-spiked sugar water in all studies. Several experiments detected an increased length of trips or a reduced number of successful trips (Gill et al. 2012; Feltham et al. 2014; Gill and Raine 2014; Stanley et al. 2015; Stanley and Raine 2016). A single study found only minor changes in foraging activity and pollen collection (Arce et al. 2016).

Immune system. Neonicotinoid exposure has been linked to increased disease and parasite susceptibility in honey bees in (semi-)field experiments (e.g., Vidau et al. 2011; Pettis et al. 2012; Alburaki et al. 2015; Dively et al. 2015). Such effects were not studied in wild bees. Because their nervous and immune systems are very similar to those of honey bees, it is possible that neonicotinoids also make wild bees more prone to disease and parasites (Wood and Goulson 2017). Fungicide effects on immune functions may also be relevant. Pettis et al. (2013) investigated the impact of collected crop pollen on Nosema ceranae prevalence in honey bees and found a correlation of infestations and pollen fungicide load.

Neglected effects

Source–sink effects. There are ecologically more complex effects resulting from intra- and interspecific interactions that have thus far been barely considered by researchers. These effects are most relevant at the population and community levels. They are not exclusive to FVIs but are especially relevant for this group (European Food Safety Authority 2015).

The FVIs can easily move between multiple in-field and off-field habitats within a landscape. Spatial movement has therefore to be considered when pesticide effects on FVI populations are investigated. Migration from semi-natural off-field habitats to pesticide-treated in-field areas could possibly result in source–sink dynamics: Individuals from a sustaining habitat migrate to a nonsustaining habitat and subsidize the sink population but also deplete the source population (Topping et al. 2015). This process can be mistaken for in-field recovery when the off-field surroundings are not considered. It has been shown in modeling studies that landscape-scale effects of pesticides cannot be sufficiently estimated using small-scale data (Topping et al. 2014, 2015). Migratory population dynamics in time and space are difficult to detect using field experiments due to limited duration and restricted spatial scale. Landscape-scale modeling approaches represent promising methods to assess source–sink effects of pesticides (Topping et al. 2015).

Indirect effects. Aside from direct effects, pesticides can also impact FVIs indirectly through trophic interactions. Habitat quality may be adversely affected by reduction or modification of food and nesting resources (Relyea and Hoverman 2006; Rohr et al. 2006). One of the main causes of FVI decline is decreased diversity and abundance of flower and nesting resources. This is caused by habitat destruction through agricultural land use practices, such as pesticide use (Goulson et al. 2015; Forister et al. 2016; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016; Ollerton 2017). Scheper et al. (2014) combined pollen load data from entomological museum collections with population trends of wild bees. Decline of preferred food plant species was identified as one of 2 main factors associated with bee species declines. Therefore, herbicide applications might reduce FVI food plant supply and consequently lead to adverse population effects. Unfortunately, there is no information available that would allow us to evaluate the relevance of indirect pesticide effect on FVIs.

Ecosystem services (pollination/biodiversity). In contrast to protection goals that were defined by authorities (United Nations 1992; European Commission 2009), there is little to no research regarding the effects of pesticide applications on FVI ecosystem services, such as pollination or biodiversity. First evidence of a direct pesticide pollination effect in a field setting was found in a semi-field cage experiment (Stanley et al. 2015). Bombus terrestris females were exposed to thiamethoxam and allowed to forage on apple trees, which subsequently reduced apple seed production. However, this is not a pollination effect in the economic sense because the number of seeds does not influence apple market value.

It is difficult to directly detect FVI diversity or population effects in field experiments, because it would take years and extensive sampling campaigns to collect the necessary data. A meta-analysis related bee species distribution monitoring data over an 18-yr period in the United Kingdom to neonicotinoid use in oilseed rape (Woodcock et al. 2016). Population persistence was negatively affected in both bee species that forage on oilseed and those that usually do not. However, the effect was 3 times stronger in oilseed rape foragers. Therefore, neonicotinoid use in a mass-flowering crop possibly caused bee species decline. However, this result of pesticide effects on FVI diversity is only correlative and cannot be connected directly to pesticide use.
After collating information on the relevant groups, their ecology and habitat, exposure to pesticides, and subsequent effects, we now critically evaluate the European exposure and effect assessment for its suitability concerning FVIs.

REGULATORY DEFICITS AND DEVELOPMENT

European risk assessment

The European pesticide risk assessment is a proactive administrative measure that should ensure the protection of non-target species as outlined in Regulation (EC) 1107/2009 (European Commission 2009). The FVIs are currently covered by risk assessment schemes for bees (European and Mediterranean Plant Protection Organization 2010a, 2010b) and NTAs (Candolfi 2001) within the framework of the terrestial ecotoxicology guidance document (Health and Consumer Protection, European Commission 2002).

However, ongoing FVI declines that are partly caused by pesticides suggest the possibility that the current risk assessment is not sufficiently protective (Godfray et al. 2015; Goulson et al. 2015; Potts et al. 2015; Intergovernmental Platform on Biodiversity and Ecosystem Services 2016). The European Food Safety Authority (EFSA) identified major shortcomings in FVI risk assessment and suggested improvements for the bee and NTA guidance documents (European Food Safety Authority 2015; European Food Safety Authority Plant Protection Products and Their Residues Panel 2012). Consequently, they drafted a new bee guidance document that should improve the risk assessment process (European Food Safety Authority 2013). This process of revising old guidance and devising a new framework is far from finished. The revised bee guidance document has yet to be ratified, and an NTA guidance document has not yet been developed. Therefore, scientific input is needed to facilitate the regulatory development.

Bee risk assessment

In the current regulatory framework, the impact of pesticides on bee species is assessed in a separate scheme, in contrast to all other FVIs (European and Mediterranean Plant Protection Organization 2010a, 2010b). Exposure and effect assessment is generally carried out as follows.

Potential exposure of bees is estimated for in-field scenarios (Table 2). At the first tier, contact contamination of individuals is evaluated by using application rates of pesticide products. Furthermore, oral exposure is considered by using data from plant residue and metabolism studies. Higher tier exposure assessment includes pesticide residue studies of relevant matrices such as dead bees, nectar, pollen, wax, or honey (European and Mediterranean Plant Protection Organization 2010a, 2010b). First-tier risk assessment requires effect testing for acute contact and oral mortality in honey bees. In the case of systemic pesticides, brood feeding tests may be necessary. In higher tier testing, several more realistic honey bee test systems can be used to refine the evaluation process if further information is required (e.g., chronic oral tests, semi-field studies using tunnel tents, or field tests; European and Mediterranean Plant Protection Organization 2010a, 2010b).

In a revision of the current guidance, the EFSA identified major deficits with regard to the ecology of FVIs (European Food Safety Authority 2013; European Food Safety Authority Plant Protection Products and Their Residues Panel 2012). The current exposure assessment does not include off-field areas, which are also FVI habitat and should therefore be considered (Table 2). Furthermore, FVI contaminations by dust from solid application as well as exposure by water sources are not incorporated. It has been noted that the entire spectrum of bee species is not well represented because the honey bee is used as the only surrogate. Other bee species’ sensitivity to pesticides is usually unknown (Arenda and Sgolastra 2014; Uhl et al. 2016). Because relative susceptibility varies for different pesticides, it is difficult to extrapolate acute toxicity data from the honey bee to wild bees (Biddinger et al. 2013; Uhl et al. 2016).

Wild bee species also have different ecological properties than the honey bee, which leads to contrasting results in complex higher tier tests (Arenda and Sgolastra 2014; Rundlöf et al. 2015; Stoner 2016). The EFSA has further stated that current semi-field and field designs generally allow for too much data variance and do not provide enough statistical power (European and Mediterranean Plant Protection Organization 2010a, 2010b).

As a reaction to the deficits in current bee risk assessment, the EFSA drafted the new bee guidance document, which includes substantial improvements (European Food Safety Authority 2013). Exposure assessment incorporates additional pesticide uptake pathways such as dust from seed treatment, guttation water, puddles, and surface water (Table 2). Aside from in-field exposure, off-field exposure is also incorporated via deposition factors for spray, granular, and seed treatment application. Residue studies should also include plant material or bees foraging on the treated crop as well as bees returning to the hive in higher tier exposure assessment (European Food Safety Authority 2013). Two additional test species were selected because of their different acute sensitivity and ecological differences that are relevant for higher tier testing: a bumble bee (B. terrestris) and a solitary bee (Osmia bicornis/ comata). In first-tier effect assessment, chronic oral and larval toxicity tests were added. The EFSA further called for modified study designs in higher tier effect assessment to decrease data variance and enhance statistical power (European Food Safety Authority 2013). Such designs include larger tunnel/field size, higher number of replicates and colonies/site, greater distance between sites, the use of sister queens in colonies, and prolonged study duration.

In spite of the extensive regulatory changes that the EFSA proposed (European Food Safety Authority 2013), deficits and open questions remain that arise from the recommendations (Table 2). The suggested exposure assessment does not include soil as a contamination source, although it is acknowledged to be relevant (Gradish et al. 2019; Sgolastra et al. 2019). The recommendations discuss honey dew and extrafloral nectar as potential exposure sources but do not provide...
information to justify their importance for FVI exposure. Regarding effect assessment, there is reasonable doubt that the proposed additional test species will decrease uncertainty. Notwithstanding the limited available database, it can be concluded that both species are usually less sensitive than the honey bee in acute toxicity tests (Uhl et al. 2016). It may still be reasonable to use these species for higher tier testing in which ecological differences influence toxicity to a greater extent (Cutler and Scott-Dupree 2014; Rundlöf et al. 2015). However, there are currently no established chronic or larval laboratory, semi-field, or field test protocols for both species. Furthermore, the ambitious study design improvements might be difficult to implement.

Several specific issues are generally not included in European risk assessment and are also not considered by the EFSA (European Food Safety Authority 2013). Neglected effects include landscape-scale source–sink effects, indirect effects through trophic interactions, and ecosystem service effects (pollination, biodiversity). These effects have not been well studied but are very relevant for the environmental safety evaluation of pesticide products. In addition, there is no assessment of effects after exposure to multiple pesticides, for example, through tank mixtures or sequential applications.

**NTA risk assessment (non-bee FVIs)**

All non-bee FVI groups are covered within the current NTA risk assessment framework (Candolfi 2001). An NTA exposure assessment is performed separately for in-field overspray and off-field spray drift scenarios, which include calculations of maximum residue levels (Table 2). The NTA effect evaluation is performed with several predatory or parasitic arthropods (e.g., Aphidius rhopalosiphi, Typhlodromus pyri). In a first-tier evaluation, acute and chronic mortality laboratory tests are conducted, whereas higher tier testing includes extended laboratory and aged pesticide residue studies, semi-field, and field experiments to study more subtle pesticide impacts under more realistic conditions, that is, lethal and sublethal effects. Four additional beneficial test species are proposed for products with a special mode of action or higher tier assessment which are used in integrated pest management (Orius laevigatus, Chrysoperla carnea, Coccinella septempunctata, and Aleochara bilineata).

Similar to bee risk assessment, the current NTA scheme needs to be adjusted to allow for a protective evaluation of pesticide impact on non-bee FVIs (Table 2). As discussed by the EFSA (European Food Safety Authority 2015), exposure caused by dust drift after sowing of pesticide-treated seeds needs to be assessed. Furthermore, there is no oral toxicity testing in the first-tier assessment, which would be relevant for FVs that consume nectar, pollen, or stem/leaf material. Moreover, non-bee FVIs are not specifically accounted for by surrogate organisms.

To alleviate shortcomings in the current guidance, the EFSA published a scientific opinion on NTA risk assessment that is the precursor of an upcoming new NTA guidance document.
(European Food Safety Authority 2015). They revised the exposure evaluation to include estimates of pesticide uptake through food (nectar, pollen, stem/leaf material) and dust as well as contamination of soil surfaces (Table 2). Furthermore, one explicit FVI species (lepidopteran larvae) has been proposed as an additional test species for effect assessment. They proposed a landscape-scale risk assessment for mobile species such as FVIs which is a major change from previous concepts. This should ensure that in-field effects do not lead to unacceptable reductions in off-field populations (European Food Safety Authority 2015). Previously neglected issues such as indirect effects, source–sink dynamics, and ecosystem service effects are also discussed. They further mention that sequential and simultaneous use of different pesticides should be included in risk assessment.

However, the EFSA did not address all deficits of the current framework and raised open questions with their recommendations for a future revised NTA guidance (European Food Safety Authority 2015; Table 2). Exposure assessment of guttation water is not included, nor is in-soil residue evaluation. Due to the multitude of different life strategies and ecological niches of non-bee FVIs, it remains unclear whether one test species will sufficiently represent this group, especially in higher tier effect assessment. Overall, the NTA scientific opinion is lacking in concrete protocols for effect and exposure assessment. A new NTA guidance document is supposed to follow up with more tangible recommendations.

Upcoming FVI guidance should make use of data-driven approaches to pesticide impact assessment. These regulatory tools allow for large-scale evaluations of FVI population by incorporating ecological information.

**FUTURE REGULATORY TOOLS**

**Trait-based analysis**

Our knowledge of the species we want to protect and their environments can enable us to develop a risk assessment that is better suited for specific groups such as FVIs. Researchers have noted that it is difficult to identify representative surrogate species for this diverse group (Uhl et al. 2016; Heard et al. 2017). Therefore, alternative approaches have to be considered that facilitate the assessment of pesticide impact on FVI communities. Ecological traits (i.e., species-specific properties) determine the breadth of the ecological niche and therefore the susceptibility of FVI populations to environmental factors. The narrower the niche, the higher the sensitivity to external stressors (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015; Hofmann et al. 2019). Therefore, it is possible to allocate FVI species to ecologically similar categories and assess their population’s vulnerability to stressors such as pesticides (Brittain and Potts 2011; Hofmann et al. 2019; Sponsler et al. 2019; Figure 4). A number of traits have been identified as relevant for population vulnerability in bee species (e.g., mobility, sociality, nesting, length of flight season/duration, and volatilism; Supplemental Data, Table S2). By combining toxicity and trait data in a modeling approach, it should be possible to make broader predictions about the consequences of pesticide use on FVI communities (Williams et al. 2010; Brittain and Potts 2011; de Palma et al. 2015; Forrest et al. 2015).

Trait-based approaches for risk assessment have already been proposed with an emphasis on the aquatic environment (Rubach et al. 2011; Van den Brink et al. 2011). The underlying concept can be easily translated to FVIs and their specific properties. A comprehensive trait database for European bees is already available for bee species vulnerability classification (Roberts et al. 2016, Trait database of European bees, unpublished data). Vulnerability models would need to be validated with extensive monitoring data. Unfortunately, for all other FVI groups there is significantly less information about the ecological parameters that influence vulnerability, and no applicable databases are available.

Similar to the effect assessment, exposure evaluation of FVI species could also be improved by analyzing trait data. The influx of pesticides into FVI habitats does not necessarily result in exposure of FVI species. However, ecological trait information can be used to assess uptake probability and identify relevant exposure pathways (Brittain and Potts 2011; de Palma et al. 2015; Sgolastra et al. 2019; Sponsler et al. 2019). A combination of trait data with pesticide application information and residue levels in habitat matrices could enable a quantitative estimation of FVI contamination through specific pathways. There are a number of traits that influence exposure potential. These ecological properties include flight activity throughout the year, daily flight activity, food plant preference (nectar), nesting (location and construction), sociality, and mobility for bees (Thompson 2001; Brittain and Potts 2011; de Palma et al. 2015; Sgolastra et al. 2019). Application dates in field cultures and pesticide persistence can be combined with the active flight period of bee species to assess the proportion of species that are potentially exposed to a specific substance (Sponsler et al. 2019). A trait-based exposure analysis could also be performed for European bee species using the database just described (Roberts et al. 2016, unpublished data). A linkage of habitat to FVI exposure is currently not possible for many relevant matrices such as soil, stem or leaf material, and non-nectar fluids. If the existing information gaps are closed, it may be possible to devise a holistic general framework that connects trait-based effect and exposure assessment, as was proposed for aquatic organisms (Rubach et al. 2011; Van den Brink et al. 2011).

**Landscape-scale modeling**

Because FVIs are mobile species, knowledge of their spatiotemporal population dynamics is required for a protective assessment of pesticide impact. This was recognized by the EFSA in their NTA scientific opinion, where they argued that a landscape-scale risk assessment should be developed (European Food Safety Authority 2015). A feasible approach is employing a model system that predicts the effects of pesticide applications on populations within the agricultural landscape (Rortais et al. 2017). The animal, landscape, and man simulation system (ALMaSS) is one possible framework that could be used to evaluate pesticide impact on predefined key species.
This system can be used to implement agent-based animal population models within a comprehensive and dynamic landscape simulation. This allows for a realistic simulation of pesticide use patterns on a spatio-temporal scale. Animal behavior parameters are modeled to predict exposure and effects at the individual level that translates into population impact. The ALMaSS suite of models can presently be applied to several arthropod, bird, and mammalian species. However, such a complex system requires detailed knowledge of the investigated landscape (land use and management) as well as extensive information about the ecology of model species (Topping et al. 2003). Landscape-scale models also need to be accompanied by FVI monitoring to validate their predictions for the use in risk assessment. There are other approaches such as the BEEHAVE model that might be easier to implement at the cost of reduced explanatory power at the landscape level (Becher et al. 2014; Rortais et al. 2017). This model was designed to simulate pesticide risk to honey bee colonies. An adapted version has been developed to provide the same functionality for bumble bees (Becher et al. 2018). It is unclear whether FVIs groups other than bees can be integrated into this framework.

Both trait-based analysis and landscape-scale modeling regulatory approaches are suitable to improve future pesticide risk assessment. The main limiting factor for their application is FVI ecological data availability.

**RESEARCH RECOMMENDATIONS**

In the present review, we identified the relevant FVI groups in the agricultural landscape as bees, flies (non-syrphids and syrphids), lepidopterans (moths and butterflies), beetles, and wasps (Orford et al. 2015; Rader et al. 2015; Grass et al. 2016). Only very limited information is available to evaluate all possible groups such as the non-bee Hymenoptera and Hemiptera. Proportions of species and individuals of the respective groups vary in different crop systems (Rader et al. 2015) and semi-natural habitats (Orford et al. 2015; Grass et al. 2016). The FVIs are flying, mobile species that live in the entire agricultural landscape. They use both farmland and non-target areas such as flower strips, field margins, and hedgerows as habitat (e.g., Marshall and Moonen 2002; Holzschuh et al. 2013; Coudrain et al. 2015; Denisow and Wrzesień 2015; Feltham et al. 2015; Tschumi et al. 2015). There is insufficient information available to assess suitable habitats of the specific FVI groups and their function in more detail. They use several compartments of these habitats to fulfill specific ecological functions such as foraging, mating, and nesting (Westrich 1990; Michener 2007). Pesticide applications on crops theoretically lead to contamination of FVI habitat compartments. Therefore, FVIs are potentially exposed to pesticides through multiple pathways. Analytic studies show that FVIs are contaminated with numerous pesticides (Mullin et al. 2010; Chauzat et al. 2011; Botías et al. 2017). There is also extensive evidence of pesticide residues in crops and non-target areas (Wood and Goulson 2017). In all habitat compartments, pesticide residues have been detected, including nectar and pollen (e.g., Mullin et al. 2010; Chauzat et al. 2011; Tosi et al. 2018), soil (Jones et al. 2014; Hvězdová et al. 2018), stems and leaves (Botías et al. 2016; Mogren and Lundgren 2016), and water sources (Samson-Robert et al. 2014; Schaafsma et al. 2015; Stehle and Schulz 2015; Schmolke et al. 2018; Wirtz et al. 2018). However, it is not possible to link habitat to FVI exposure with the current knowledge base. There is a lack of information regarding the exposure of all non-bee FVI groups. The FVIs are affected by many pesticides, most notably neonicotinoid insecticides, at environmentally realistic doses. Bee (semi-)field studies found adverse effects on ecologically relevant parameters such as reproduction, foraging, and immune functions (e.g., Gill et al. 2012; Whitehorn et al. 2012; Alburaki et al. 2015; Dively et al. 2015; Feltham et al. 2015; Rundlöf et al. 2015). Furthermore, there are ecologically important...
effects at the population/community level that have been neglected so far by research such as source–sink effects, indirect effects, and effects on the ecosystem services pollination and biodiversity (European Food Safety Authority 2015).

The existing and proposed future risk assessment frameworks contain some deficits regarding the exposure and effect evaluation for FVs. Both the current bee and NTA risk assessments (Candolfi et al. 2001; European and Mediterranean Plant Protection Organization 2010a, 2010b) fail to cover the specific ecological properties of FVI species. The EFSA has drafted new regulatory documents that improve the risk assessment process (European Food Safety Authority 2013, 2015). However, there are still unaddressed issues and uncertainties that need to be resolved to achieve a protective risk assessment scheme for FVs. Data-driven tools can help to improve FVI risk assessment by using ecological information. Trait data can be used to determine their exposure to pesticides and the vulnerability of FVI populations to stressors (Rubach et al. 2011; Van den Brink et al. 2011). This information could be combined with toxicity, pesticide application, and residue data to assess pesticide impact on FVI communities in a connected framework. Another promising approach is landscape-scale modeling, which allows for an evaluation of the pesticide exposure of FVI populations and subsequent effects, in space and time (Topping et al. 2003; Rortais et al. 2017). Both approaches require comprehensive databases (currently not sufficiently available) that include FVI species traits, landscape composition, land use, pesticide toxicity, and residues in relevant matrices.

Throughout the present review, we highlighted knowledge gaps that need to be closed to better understand FVs and assess the effects of pesticide applications in their habitat. Therefore, we call for general research on the following subjects: 1) identification of all relevant FVI groups aside from bee species; 2) study of the ecology and habitat of all FVI groups; 3) implementation of extensive FVI population monitoring campaigns to determine the threat level of specific groups; 4) creation of a comprehensive FVI ecological trait database; 5) determination of FVI habitat exposure with consideration of relevant matrices and creation of a pesticide residue database; 6) linkage of habitat to FVI exposure with special regard to non-bee FVs; 7) assessment of pesticide effects with a focus on population-relevant parameters, especially for non-bee FVs; 8) investigation of neglected source–sink, indirect, and ecosystem service effects (pollination, biodiversity); and 9) development and advancement of suitable trait-based approaches for a impact assessment on the landscape scale.

Aside from this scientific input, there is a need for regulatory decision-making processes to move away from arbitrary conservative assumptions and overcomplicated risk assessment schemes toward a more substantiated and holistic approach that incorporates large-scale evaluation methods and utilizes ecological information.

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