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Protection of wild pollinators in the pesticide risk assessment and management

Final Report

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Final Report

by

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Abstract

In the European agricultural landscape pesticides are applied to crops to control pests and weeds and increase yield. However, non-target species are inevitably exposed to these chemicals, too. Flower-visiting insects (FVI) are a functional group of insects that may be particularly exposed to pesticides due to their mobility and foraging activity in crop and non-crop areas. Furthermore, there is growing evidence that global FVI numbers are decreasing and pesticides are proposed as one of the causes for this development. Flower-visiting insect species not only contribute substantially to biodiversity and aesthetic value of the agricultural landscape but many FVIs are also vital pollinators of crops and wild plants. To preserve the ecological status of the agri-environment and ensure stable crop pollination it is necessary to assess and manage the risk of pesticides towards FVIs.

In this research and development project, we reviewed the scientific literature on FVIs regarding their ecology, exposure to pesticide, subsequent effects and risk mitigation measures. Furthermore, we examined existing and drafted regulatory documents. Comparing scientific state of knowledge and regulatory status quo, we identified general deficits in current FVI risk assessment. Moreover, we determined the relevant taxonomic groups of FVIs and characterised their habitat. These taxonomic groups were divided into ecologically similar categories whose population vulnerability was assessed using ecological trait data. Thereafter, we developed exposure scenarios of FVI habitats, identified exposure-relevant traits and summarised the scientific knowledge on pesticide residues in FVI individuals and their habitat matrices. Furthermore, we compiled estimation methods for all relevant exposure scenarios. We collated evidence of pesticide effects on FVIs from studies of different complexity and described the bandwidth of lethal and sublethal effects. Moreover, FVI species sensitivity towards pesticides was compared and the selection of suitable surrogate species was discussed. Using this information, recommendations for the pesticide risk assessment scheme on FVIs were derived. Moreover, potential risk mitigation measures to reduce pesticide exposure and to promote FVI populations in agricultural regions were evaluated according to their efficiency, feasibility and acceptability by farmers. Based on this analysis recommendations for the improvement of potential risk management options were developed. Additionally an overview of possible opportunities for funding of risk mitigation measures on EU-level (e.g. greening programme) and exemplarily for selected agri-environment programmes on national level, is given.

Finally, we identified knowledge gaps in all chapters and highlighted research opportunities to further deepen our understanding of pesticides effects on FVIs and improve the existing regulatory pesticide risk assessment.

Kurzbeschreibung

In Agrarlandschaften werden mit dem Ziel einer Ertragssteigerung Pestizide in Kulturpflanzen angewendet, um sogenannte Schadorganismen zu kontrollieren. Allerdings werden dabei zwangsweise auch Nicht-Zielarten gegenüber diesen Chemikalien exponiert. Blütenbesuchende Insekten (*Flower-visiting insects* (FVI)) stellen eine funktionelle Gruppe von Insekten dar, die auf Grund ihrer Mobilität und ihrer Fouragier-Aktivität sowohl auf behandelten Anbauflächen als auch auf benachbarten Flächen besonders gegenüber Pestiziden exponiert sind. Zudem wächst die Beweislast, dass FVI-Bestände weltweit abnehmen. Als einer der Gründe für diese Entwicklung wird der Einsatz von Pestiziden diskutiert. Blütenbesuchende Insektenarten tragen nicht nur zur Biodiversität und zum ästhetischen Wert einer Agrarlandschaft bei, sondern sind darüber hinaus auch wichtige Bestäuber von Kultur- und Wildpflanzen. Um den ökologischen Wert von Agrarökosystemen zu erhalten und eine stabile Bestäubung sicher zu stellen, sind daher sowohl eine Bewertung als auch ein Management des pestizidbedingten Risikos für diese Insektengruppe notwendig.

In diesem Forschungs- und Entwicklungsprojekt haben wir die wissenschaftliche Literatur zu FVI bezogen auf ihre Ökologie, toxikologische Sensitivität und Exposition gegenüber Pflanzenschutzmitteln sowie auf potentielle Risikominderungsmaßnahmen betrachtet und analysiert. Durch den Vergleich des aktuellen Stands der Wissenschaft mit dem regulatorischen Status Quo konnten wir generelle Defizite in der aktuellen FVI-Risikobewertung aufzeigen. Zudem identifizierten wir relevante Insektengruppen innerhalb der FVIs und charakterisierten ihre Habitate. Die taxonomischen Gruppen wurden in ökologische Kategorien eingeteilt und ihre Vulnerabilität wurde anhand von ökologischen Merkmalsdaten beurteilt. Darauf aufbauend entwickelten wir Expositionsszenarien für FVI-Habitate, identifizierten expositionsrelevante Merkmale und fassten den wissenschaftlichen Kenntnisstand zu Pestizidrückständen in FVI-Individuen und in ihren Habitaten zusammen. Außerdem stellten wir Abschätzungsmethoden für alle relevanten Expositionsszenarien und Nachweise für Effekte von Pflanzenschutzmitteln aus Studien mit unterschiedlicher Komplexität zusammen und beschrieben die Bandbreite der letalen und subletalen Effekte. Zudem wurden die Empfindlichkeiten von blütenbesuchenden Insektenarten gegenüber Pestiziden miteinander verglichen und es wurde eine Auswahl von möglichen Stellvertreterarten diskutiert.

Anhand dieser Informationen wurden Empfehlungen für ein Risikobewertungsschema für FVIs abgeleitet. Zudem wurden mögliche Risikominderungsmaßnahmen zur Reduzierung der Exposition gegenüber Pestiziden und zur Förderung von FVI-Populationen in der Agrarlandschaft beschrieben und es wurden ihre Effektivität, ihre Durchführbarkeit und die Akzeptanz der Maßnahmen durch Landwirte beurteilt. Basierend auf dieser Analyse wurden Empfehlungen für die Weiterentwicklung von Risikomanagementmaßnahmen entwickelt. Zusätzlich wurde ein Überblick über Fördermöglichkeiten von Risikominderungsmaßnahmen zum einem auf EU-Ebene (z.B. im Rahmen des *Greening*-Programms) und zum anderen auf nationaler Ebene am Beispiel ausgewählter Agrar-Umwelt-Programme erstellt.

Abschließend wurden bestehende Wissenslücken identifiziert und es wurden Vorschläge für weitere Forschung skizziert, die einen Beitrag zur Vertiefung unseres Verständnis der Effekte von Pestiziden auf FVIs und zur Verbesserung existierender regulatorischer Risikobewertungsverfahren leisten soll.

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AR	Application rate
CI	Confidence interval
EFA	Ecological focus area
EFSA	European Food Safety Authority
ERA	Ecological risk assessment
EU	European Union
FVI	Flower-visiting insects
HD5	Hazardous dose
ITD	Intertegular distance
LOD	Limit of detection
LOQ	Limit of quantification
NTA	Non-target arthropod
PER	Proboscis extension reflex
PIED	Predicted initial environmental dose
RUD	Residue unit dose
SD	Standard deviation
SPG	Specific protection goal
SSD	Species sensitivity distribution

Summary

Introduction

Pesticides are applied in crops to reduce pest pressure and thus increase yield. However, not only pests but also non-target species are exposed to these chemicals. Through multiple pathways, such as root uptake and subsequent translocation in the plant or direct overspray, flowers of crops and weeds may be contaminated with pesticides that might consequently expose flower-visiting insects (FVIs). Many FVIs are vital pollinators of crops and wild plants in the European agricultural landscape. Non-bee FVIs include flies, beetles, moths, butterflies, wasps and ants which include many important pollinators (Rader et al. 2015). Plant pollination is a central ecosystem service since 35% of global agricultural production is related to crops that increase yield when pollinated by animals (Klein et al. 2007). The global economic value of animal pollination was estimated to be €153 billion (Gallai et al. 2009). However, biotic pollination is not just economically important but also essential for the preservation of native flora since 85% of all flowering plants are pollinated by animals (Ollerton et al. 2011). Flowering agricultural crops and wild plants are mostly pollinated by insects (Klein et al. 2007; Ollerton et al. 2011).

FVIs are protected under several guidelines, declarations and regulations on a European and worldwide level. Most recently, at the 13th meeting of the Conference of the Parties to the Convention on Biological Diversity (CBD) the "Cancun Declaration on Mainstreaming the Conservation and Sustainable Use of Biodiversity for Well-Being" was passed (United Nations 2016). Member states commit to take effective measures to counteract biodiversity loss. A guidance for the agricultural sector that lists actions to advance this goal includes the effective management and conservation of pollinators. Moreover, a "Coalition of the Willing on Pollinators" was created by 12 EU countries which commit to pollinator protection (Coalition of the Willing on Pollinators 2016).

There is growing evidence that global FVI numbers are decreasing because of environmental change related to factors such as climate change, habitat loss and fragmentation, environmental pollution and pesticide use (Goulson et al. 2015). This development has been most prominent in bee species. In Germany 53% of bee species are red listed (Westrich et al. 2011), in some European countries even up to 65% (Patiny et al. 2009). Since the last century the USA and Europe are experiencing substantial losses of domestic honey bee (*Apis mellifera*) hives and simultaneous decline in wild bee diversity (Natural Research Council 2006; van Engelsdorp et al. 2008; Potts et al. 2010; Goulson et al. 2015; Potts et al. 2015). A population decline of butterfly, moth and syrphid fly species is also recognised in the EU (EASAC 2015; Gilburn et al. 2015; Potts et al. 2015; Forister et al. 2016). Pesticide effects on FVIs are most noticeable in bee species, especially honey bees since they are important pollinators and there is an economic interest in preserving viable populations (Klein et al. 2007; Gallai et al. 2009). However, there are many other flower-visiting taxa that might be exposed to pesticides in the agricultural landscape and consequently be affected (Godfray et al. 2014; Gilburn et al. 2015; Godfray et al. 2015). Hence, the terms "pollinator" and "flower-visiting insect" are to be distinguished and the whole community of FVI species should be addressed when investigating the impact of agricultural pesticide use. In the context of this report the term FVI is referring to insect taxa that forage on flower resources such as nectar and pollen in at least one life stage.

Regulatory development

The currently valid guidance document on Terrestrial Ecotoxicology under Council Directive 91/414/EEC (SANCO 2002) refers to a protocol of the European and Mediterranean Plant Protection Organization for bee risk assessment schemes (OEEP/EPPO 2010a, b). The honey bee has been used as the sole surrogate organism for bees which has been criticised due to substantial ecological differences to most other species (Arena & Sgolastra 2014; Rundlöf et al. 2015; Stoner 2016). Furthermore, several deficits have been identified regarding for example the inclusion of relevant exposure

scenarios or insufficient statistical power of field and semi-field test designs (EFSA PPR Panel 2012; EFSA 2013). All non-bee FVIs are currently covered by NTA risk assessment which refers to results of the Escort 2 workshop (Candolfi et al. 2001). This scheme also shows clear deficits as it does not include any FVI test organisms and does not consider oral uptake in effect assessment.

EFSA recognised these deficits and drafted a bee guidance document (EFSA 2013) and NTA scientific opinion (EFSA 2015) that incorporate substantial improvements compared to the current risk assessment process regarding FVIs. However, one remaining major problem is that bees and all other FVI species are subject to different risk assessment schemes. This complicates the implementation of an effective risk assessment process for the whole group of FVIs. Test species selection and representativeness for FVI communities is also an important issue. The honey bee's representativeness for bee species let alone other FVI groups is still questionable due to its physiological and ecological properties. (Cutler et al. 2014; Rundlöf et al. 2015). The preliminary selection of the additional test species *Bombus terrestris* and *Osmia bicornis* is also disputable since there is uncertainty concerning their suitability as surrogate species and their sensitivity as test organisms. However, due to ecological differences, especially concerning their sociality, the two suggested additional test species might be suitable to improve higher tier risk assessment (Cutler et al. 2014; Rundlöf et al. 2015). Introducing a lepidopteran surrogate test species for non-bee FVIs is a reasonable measure but this additional test species should be selected with great care. It is unclear if one species is enough to represent this ecologically diverse group since information on sensitivity of non-bee FVI species is scarce. Furthermore, exposure scenarios for such a diverse group as FVIs are hard to define since there is no comprehensive review of the exposure risk throughout FVI taxa and their life history up until now. These exposure pathways should also be considered in effect assessment (e.g. oral pesticide effects on herbivore FVI larval stages are not incorporated). Furthermore, adequate and feasible laboratory and (semi-)field test protocols for the proposed additional test species need to be developed and validated.

Overall, the recent EFSA revisions of bee (EFSA PPR Panel 2012; EFSA 2013) and NTA risk assessment (EFSA 2015) still do not sufficiently incorporate the ecological properties of FVI species. Furthermore, it is necessary to address more complex issues such as indirect effects of herbicides on FVIs through food depletion, landscape-scale population source-sink effects or effects on FVI ecosystem services (e.g. pollination, biodiversity). To achieve a protective risk assessment for FVIs, ecological characteristics of the different taxa of this group need to be addressed regarding exposure and effects. Furthermore, a set of effect measures needs to be defined, as well as, acceptable effect levels. When these prerequisites are met, more protective risk management measures can be formulated.

Taxonomic groups of flower-visiting insects

In the scientific literature the main groups of FVI species are often identified as bees, hover flies and lepidopterans (Winfree et al. 2011). However, more recent studies found that FVI communities in the agricultural landscape are far more diverse and the current definition of relevant groups seems incomplete. Grass et al. (2016) investigated flower visitations of insects in wildflower plantings situated in the mid-German agricultural landscape. They found that aside from bees and hover flies flowers were visited by a diverse community of other insect taxa. In fact, non-bee/non-hover fly insects made up half of the visiting individuals and 75% of FVI species. Furthermore, non-hover fly Diptera were by far the largest portion of visiting insect species. In contrast, butterflies only made up a small share of FVI abundance whereas flower visitation by beetles and non-hover fly Diptera in terms of individual numbers was comparable to that of honey bees. Further meta-analyses in crops and non-target areas provide additional evidence that FVI communities are more diverse than previously assumed (Orford et al. 2015; Rader et al. 2015).

With the current data we can define the relevant FVI groups in crops and their semi-natural surroundings as bees, flies (non-syrphids and syrphids), moths and butterflies and beetles.

Proportions of species and individuals of the respective groups are rather variable in different crop systems (Rader et al. 2015) and semi-natural habitats (Grass et al. 2016). For non-bee Hymenoptera and Hemiptera there is not enough information available to make statements on their relevance as FVIs. Despite clear identification of major FVI groups sufficient information to evaluate the risk of pesticides is only available for bees and Lepidoptera. There are eminent knowledge gaps regarding the ecological information for flies and beetles. Therefore this report is focusing mainly on bees and Lepidoptera.

Habitats of flower-visiting insects

The agricultural landscape provides diverse habitats for FVIs which can be categorised into three general types: in-crop/in-field (crop plantings), off-crop/in-field (managed flower strips) and off-crop/off-field (semi-natural field edge structures). These areas differ in many aspects such as structural composition, plant species inventory/diversity, spacial and temporal food resource availability, natural enemies or anthropogenic stress such as pesticide input. Cropping systems differ in their suitability as FVI habitats for bee species due the attractiveness of the crop. However, for other FVI species it is more difficult to evaluate crops as habitats due to data scarcity. For FVI groups such as Lepidoptera or beetles crops can be a habitat not only for adults but for their herbivorous larval stages. Furthermore, non-attractive crops areas might still be FVI habitats if there is undergrowth of crop-associated wild plants (Storkey & Westbury 2007; Balmer et al. 2014; Manandhar & Wright 2016). Moreover, even without any flowering plants present in-crop areas can still provide habitat functions for FVIs (e.g. nesting area for bees, flies and beetles or temporary refuge for all flying FVI life stages). There are multiple types of field-adjacent structures which differ in plant species inventory and habitat suitability for FVIs species. They are the major type of semi-natural habitat in intensely managed agricultural areas and provide multiple habitat function for FVI species (Marshall & Moonen 2002). Flower strips as agri-environmental management measures are designed to function as suitable habitats for FVI species which was demonstrated in scientific studies (e.g. Garibaldi et al. (2014); Feltham et al. (2015)).

Within the agricultural landscape crops as well as field edge structures and flower strips are habitats of FVI species. FVI species use different parts of their habitats to fulfill specific functions throughout their life cycle. Depending on the ecological attributes of FVI species relevant compartments of habitats (e.g. airspace, flower, stem/leaves, soil, water reservoirs) used by these species throughout their life cycle vary in time and space.

Ecological categories and population vulnerability

Ecological attributes (i.e. traits) determine the susceptibility of FVI populations to environmental change (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015). Therefore, it is possible to allocate FVI species to ecologically similar categories and assess their vulnerability to stressor such as pesticides.

Bees (Hymenoptera: Apoidea: Anthophila) are a monophyletic group of more than 1900 European species (Nieto et al. 2014). Aside from the well-known western honey bee *Apis mellifera* there are a multitude of ecologically quite variable wild bee species in Europe. Some species are for example eusocial, i.e. live in colonies or aggregations, but most species are solitary. Additionally, there are many parasitic species that exploit their host to feed and tend to their offspring (Michener 2007; Goulson 2010). There are several nesting strategies in bee species: Most species burrow into the soil to build their nest but others also occupy pre-existing cavities in soil or deadwood or construct nests from collected material (Michener 2007).

Several traits (mobility, sociality, nesting, lecty, flight season/duration, voltinism) were analysed using data from a comprehensive database for all European bee species (Roberts et al. (unpublished) to allocate bee species to categories of similar ecological vulnerability and chose representative species from these groups. European bee species were assigned to seven relevant categories considering three ecological traits (mobility, lecty, sociality). Afterwards focal species were identified from the respective categories, considering a wide distribution in Europe and representative trait combinations with special emphasis on a representative flight activity throughout the year. Furthermore, the three major traits identified for categorisation of bee species plus flight season/length were used for the qualitative evaluation of ecological vulnerability. Out of the seven categories three were assessed as highly vulnerable, three as medium vulnerable and only one as least vulnerable. These vulnerability groups should be confirmed by actual population records and ecological studies. Only mid- to long-term monitoring can show if populations of designated vulnerable categories are actually decreasing. Unfortunately, these data are scarce for most European bee species. Such data should be collected in a Europe-wide population monitoring of bees.

Moths and butterflies (Lepidoptera) are a common and species rich phytophagous insect group. Although Lepidoptera are one of the most studied arthropod groups, the majority of Lepidoptera research has focused on diurnal butterflies (New 2004), which represent only 10% of the Lepidoptera species (Shields 1989). The remaining species are classified as moths and have often crepuscular and nocturnal lifestyles. Larvae (i.e. caterpillars) of most lepidopteran species are herbivores and feed on plant material such as leaves, roots, flowers, seeds, or fruits (Scoble 1995). As some Lepidoptera species also feed on crops during their caterpillar stage, they have been classified as agricultural pests. However, the majority of Lepidoptera species feed on non-crop plants (Ebert 1994; Scoble 1995). In their adult stage, numerous Lepidoptera species feed on nectar and occasionally on pollen (Scoble 1995).

Several ecological attributes of lepidopterans (especially butterflies) have been identified to determine species' vulnerabilities to changes in environment and climate (Aguirre-Gutierrez et al. 2016, Eskildsen et al. 2015). These traits include mobility of adults, habitat specialisation, host plant specialisation of the caterpillars and overwintering stage. Furthermore, other characteristics, such as the time and length of the adult flight period or the preferred growing conditions of host plants regarding nitrogen-input (eutrophication) might be also of relevance (Franzen & Johannesson 2007; Kuussaari et al. 2007; Barbaro & van Halder 2009; Aguirre-Gutierrez et al. 2016). Hence, these traits can be suitable to characterise the ecological vulnerability of Lepidoptera species. However, assigning butterfly species to ecological categories is not feasible at the moment since a comprehensive database of European moths or butterfly species traits is not available. Since there are no definite ecological categories it is difficult to define focal species. However, there are several lepidopteran species that have been used in ecotoxicological studies and might also be applicable as focal species due to their wide distribution.

Ecological attributes determine the vulnerability of FVI populations towards stressors and there already is a comprehensive database for many traits of bee species to derive ecologically similar groups and chose focal species. These groups can also be evaluated regarding their vulnerability. However, theoretically vulnerable groups should not be mistaken for actually threatened groups. Actual threat still needs to be determined by population monitoring. For lepidopterans some traits have been identified that ecologically distinguish groups and might give information about their populations' vulnerability but there is not enough data to conclusively define similar groups or chose focal species. There is a need for further studies on the ecological attributes of moths, butterflies and other non-bee FVI species.

Exposure of habitats

There are several processes that transport pesticides into FVI habitats. These can generally be assigned to two distinct groups: Primary processes characterise the intended application of pesticides to a crop (in-field habitat). These methods include spray application of mostly non-systemic pesticides and the application as a solid formulation (seed treatment, granules), stem injection or irrigation of systemic pesticides. Furthermore, a fraction of these applied pesticides is unintentionally redirected into off-field habitats by spray drift, field edge overspray, dust dispersion/drift and run-off. A wide array of habitat compartments and matrices are reached by primary and secondary processes including the airspace, pollen and nectar of flowers, stem and leaves of plants, guttation water, extrafloral nectaries, honeydew, small ephemeral water bodies (i.e. puddles), larger water bodies (river, lakes) and soil in in-field and off-field habitats. These processes can also lead to combined exposure of FVI habitats. Furthermore, due to the high soil persistence of some systemic pesticides (e.g. neonicotinoids) succeeding attractive crops (and weeds) might be exposed to accumulated residues from applications in previous cropping seasons (Goulson 2013b; Bonmatin et al. 2015).

Exposure of flower-visiting insects

The influx of pesticides into FVI habitats is not necessarily resulting in exposure of FVI species. However, ecological trait information can be used to assess uptake probability through different habitat matrices and identify relevant exposure pathways. Unfortunately, comprehensive trait information is only available for bee species which does not allow for a detailed analysis of other FVI groups. In bee species there are several traits that have been identified to influence the extent of their exposure. The annual flight period of most wild bee species coincides with crop growing season and therefore with pesticide applications in crops. Depending on the crop bee species might therefore be exposed to a wide variety of pesticides. Additionally there is potential exposure to pesticides that are applied outside the active flight period due to uptake by foraging of persistent compounds in soil and plant material (Fantke & Juraske 2013; Goulson 2013b; Bonmatin et al. 2015). Apart from their annual activity window bees do also differ in their diel activity patterns. Therefore, mitigation measures such as delaying pesticide application into evening hours to minimise honey bee exposure might maximise bumble bee exposure. Oligolectic bee species may be preferentially exposed from their narrow range of food plants whereas polylectic bee species may be exposed to pesticide residues from a wide flower spectrum. The nesting trait of bees is also relevant when assessing their exposure potential. Since the majority of bee species (64%) dig into the ground to build their nests ("ground excavator"), soil exposure should be recognised as a critical pathway. However, some bee species may also be exposed by plant material collected for nest building from different in-crop weeds and off-crop non-target plants (Westrich 1990). Furthermore, social bee species might have a higher probability to be exposed to pesticides than solitary bees due to the sheer number of foragers and the amount of collected pollen from a wide array of plants (Brittain & Potts 2011). The assessment of exposure-relevant traits in Lepidoptera is more difficult than in bees since there is no comprehensive database of butterfly traits available. Therefore, ecological attributes of lepidopteran species can be listed but their relevance for exposure incidence in butterfly and moth communities need to be evaluated in follow-up studies.

Investigations of pesticide residues levels in FVI individuals are indispensable to quantitatively assess pesticide exposure. Unfortunately, these data are only available for bees at the moment. Data for other FVI groups such as lepidopterans, flies and beetles should be collected to allow for adequate exposure assessment. Bees are exposed to a plethora of pesticides: Studies that analysed pesticide residues in honey bee brood, pollen and honey detected up to 98 substances and degradates (Chauzat 2006; Mullin et al. 2010). In a review of the environmental impact of neonicotinoids Wood & Goulson (2017) compiled expected residues of three neonicotinoids in pollen and nectar of selected crops that were calculated by EFSA. Pollen doses were consistently higher than nectar doses and residue levels generally fluctuate between crops. The authors concluded that several parameters such as dose and mode of treatment, studied crop, season, location, soil type, weather and time of day samples were collected influence neonicotinoid doses in both matrices. There is, however, still a small data basis on residues in pollen and nectar collected by wild bee species.

Bees and FVIs in general collect pollen and nectar from a wide variety of plants in off-crop habitats. However, several more recent studies show that vegetation in non-target areas (field margins adjacent to fields) is often contaminated with pesticides, too (Wood & Goulson 2017). Recent studies report mean levels of neonicotinoids in pollen from <0.4 to 14.8 ng/g and nectar from <0.1 to 1.5 ng/g. These residue levels are comparable to residue levels found in seed-treated crops when flowering (Wood & Goulson 2017). To link exposure of wild plant pollen to bee exposure it is necessary to analyse bee-collected pollen sources. In several other studies with free-flying honey bees as pollen collectors there are noticeable trends: The highest levels of residues are found when a large proportion of crop pollen is collected. Conversely, when wildflower proportions in pollen are higher neonicotinoid residue doses are lower (Wood & Goulson 2017). There are very few studies which investigated pollen collected by other bee species which found similar levels of insecticides. In general, all these studies show that high acute doses in pollen and nectar coincide with the bloom of FVI-attractive crops. However, chronic exposure of species with a long active flight period such as honey bees might be driven by wildflower foraging (Botías et al. 2015; Wood & Goulson 2017). Wild bees may be exposed to a wide variety of pesticides when foraging in non-target areas as shown by Long & Krupke (2016).

Since the majority of European bees species nest in the soil, pesticide exposure by soil contact may be an important exposure pathway. Soil exposure may also be relevant for soil-dwelling life stages of other FVI groups (e.g. fly and beetle larvae). There have been several studies which measured neonicotinoid residues in agricultural soils which show a range of 0.4-13.3 ng/g for imidacloprid, clothianidin or thiamethoxam levels in different cultures with varying previous crops. To assess the pesticide exposure of FVIs from soil it is not only important to know (peak) concentrations but also the persistence in the soil matrix which range from several days to years for neonicotinoids (Goulson 2013a). There are a few studies that show accumulation or exposure by application of previous years even if these substances are not applied anymore (Bonmatin et al. 2005; Goulson 2013a; Jones et al. 2014). However, it is difficult to link soil exposure to pesticides to FVI exposure through soil contact. Systemic pesticides are designed to be taken up by crops from the soil. However, wild plants can also take up systemic pesticides. Generally, FVI species might be exposed to pesticide residues in or on plant material (e.g. herbivore life stages of lepidopterans and beetles or bees collecting nest materials). A couple of studies have shown residues in wild plant stem or leaves at comparable levels to crops (Botías et al. 2016; Mogren & Lundgren 2016). Moreover, flower-visiting insects may also be exposed to toxicologically relevant pesticide concentrations when consuming water from small ephemeral water bodies (i.e. puddles) in crops (Samson-Robert 2014; Schaafsma et al. 2015). Furthermore, pesticide concentrations in guttation water that is secreted by some plants but can be up to four to five orders of magnitude higher than concentrations in nectar (Godfray et al. 2014). However, a clear link of pesticide residues in guttation droplets and pesticide uptake of bees remains to be provided (Wood & Goulson 2017).

To estimate exposure of FVI species through the multitude of relevant pathways, conceptual frameworks have been proposed in recently drafted guidance documents and scientific opinions (EFSA 2013, 2015; SANCO 2014). The proposed estimation procedures can be adapted and improved with results from scientific studies in some cases. In other cases there is no adequate theoretical framework currently available to assess exposure and therefore chemical residue analysis of relevant matrices should be used instead. Since FVIs are mobile species, knowledge about their spatio-temporal pattern of exposure is required for the assessment of possible risks to populations. This was also recognised by EFSA in their NTA scientific opinion (EFSA 2015). To adequately assess pesticide exposure of other FVI groups (e.g. lepidopterans, flies and beetles) from nectar and pollen feeding, field-adjacent food uptake rates of their relevant life stages should be collected.

There is extensive evidence that bees are exposed to pesticides not only through direct overspray or spray/dust drift but also by consuming contaminated food item such as pollen and nectar or water. Furthermore, bees can be exposed while collecting nesting materials or digging their nests in the soil. These exposure pathways are probably also valid for life stages of other FVIs who consume pollen or nectar (e.g. lepidopterans, beetles, flies), stem or leaf material (e.g. lepidopterans, beetles), water, collect plant materials for nest building or nest in the soil (e.g. beetles, flies). There is some ecological trait information for bees which allows for evaluation of their exposure probability to specific habitat matrices. However, this database needs to be expanded for bees and established for other FVI groups. Furthermore, pesticide residue data in all relevant matrices needs to be collected (especially in off-field non-target plants) to quantitatively assess FVI exposure and create as well as validate adequate exposure models. Landscape-scale modelling can be a valuable tool to evaluate FVI exposure in space and time.

Pesticide effects on flower-visiting insects

Since the honey bee is included as a test organism in European pesticides risk assessment there are acute toxicity data for all registered pesticides. However, other bee species' sensitivity is usually unknown which makes it difficult to establish the honey bee as a surrogate organism for risk assessment of wild bees or even other FVIs. Arena & Sgolastra (2014) analysed the available literature and found a bridging factor of 10 on top of a honey bee LD50 to cover wild bee species sensitivity in 95% of all cases. Uhl et al. (2016) assessed acute toxicity of dimethoate towards several European wild bee species and concluded that this bridging factor would allow for a protective assessment of the risks associated with dimethoate when applying the SSD concept. However, they also noted that relative susceptibility varies for different pesticides and that the bridging factor might need to be adapted for other pesticides. Furthermore, there is much variance when comparing acute sensitivity of wild bee species and the honey towards different pesticides (Uhl et al. in prep.). Consequently, there is not only a need to screen more species for their sensitivity towards one standard substance (e.g. dimethoate) but also to test single wild bee species with an array of pesticides. Furthermore, mixture toxicity has also been investigated in laboratory studies with wild bees. Synergistic and antagonistic effects of pesticide combinations have been shown in the wild bee species *B. terrestris* and *O. bicornis* and should therefore be considered in FVI effect assessment (Sgolastra et al. 2016; Robinson et al. 2017).

There have been several studies investigating toxic effects of pesticides below lethal doses (sublethal effects) on wild bees species, mostly bumble bees. At environmentally realistic doses effects on reproduction and foraging on *Bombus* species were detected that were measured in parameters such as oocyte development, oviposition, nest-building activity, colony weight gain, worker life span, proboscis extension reflex, feeding or flower visitation (e.g. Morandin & Winston 2003; Laycock & Cresswell 2013; Scholer & Krischik 2014; Stanley et al. 2015b; Baron et al. 2017). Furthermore, there have been contrasting results regarding combination effects of pesticides and parasites (Fauser-Misslin et al. 2014; Piiroinen et al. 2016; Piiroinen & Goulson 2016; Fauser et al. 2017). Aside from bumble bees sublethal laboratory pesticide effects have also been studied in a few experiments for other wild bee species in parameters such as larval development, egg production, overwintering performance, offspring sex ratio or hatching rate (Abbott et al. 2008; Sandrock et al. 2014).

In 2013 the European Commission restricted the neonicotinoid compounds imidacloprid, clothianidin and thiamethoxam in use because of high acute risks for bees. Since then several complex field and semi-field studies have been carried out to further the understanding of neonicotinoid effects on bees, honey bee and wild bee species, in the agricultural landscape. Within this more realistic setup ecological differences between honey bee and wild bee species are assumed to result in diverging responses to pesticides (Stoner 2016; Wood & Goulson 2017). The following studies established the impact of pesticides on different parameters that are crucial for the maintenance of stable wild bee populations. The better part of experiments that investigated reproduction effects found negative impacts on bumble bee colony development (e.g. Cutler & Scott-Dupree 2014; Moffat et al. 2015; Moffat et al. 2016; Ellis et al. 2017) or solitary bee brood cell construction (Rundlöf et al 2015). However, two large field studies did not identify impact on reproduction in either *B. terrestris* or *O. bicornis* (Peters et al. 2016; Sterk et al. 2016). Pesticide treatment also affected foraging in wild bee field and semi-field experiments. Several studies showed a general pattern of increased number and duration of foraging trips but decreased foraging efficiency (e.g. Gill et al. 2012; Gill & Raine 2014; Feltham et al. 2014; Stanley et al. 2015a; Stanley & Raine 2016). Only one recent experiment by Arce et al. (2016) found just minor changes in foraging activity and pollen collection. Effects on the immune system of honey bees (increased disease and parasite susceptibility) have been shown in multiple studies (e.g. Alburaki et al. 2015, Dively et al. 2015, Pettis et al. 2012, Vidau et al. 2011). Such effects were to this day not studied in wild bees in field scenarios.

There are several laboratory and field investigations of the abovementioned effects. However, there are a couple of relevant effects that have rarely been assessed or not at all. Effects on pesticide formulation adjuvants have only been evaluated in a few more recent studies which found toxicity of these substances to be similar to the actual active ingredient and also synergistic interaction with it (Ciarlo et al. 2012; Zhu et al. 2014; Mullin et al. 2015; Fine et al. 2017). Furthermore, indirect effects on FVI populations such as reduced habitat quality through reduction of food and nesting resources were rarely investigated so far. Decreased diversity and quantity of flower resources caused by habitat destruction and agricultural land use practices are suspected as main factor of causing bee declines (Müller et al. 2006). Moreover, the impact of pesticides on population dynamics within a landscape has rarely been evaluated. Source-sink effects on FVI (meta-)populations have only been addressed in landscape-scale simulation studies by Topping et al. (2014; 2015). In contrast to protection goals that were defined by EFSA (2015), effects of pesticide exposure on ecosystem services such as pollination and biodiversity remain to be adequately investigated. Stanley et al. (2015a) found first evidence of reduced pollination of apple trees by *B. terrestris* in a semi-field experiment. Studies on biodiversity effects are even more scarce.

There is a much smaller number of studies concerning the impact of pesticides on moths and butterflies than on bees. Studies on direct toxic effects (mortality) of insecticides on Lepidoptera have focused on the herbivore larval stage (i.e. caterpillars) (e.g. de Jong et al. 2008; Hoang et al. 2011; Han et al. 2012; Hahn et al. 2015b; Pecenka & Lundgren 2015). However, in some studies, direct toxic effects of insecticides on adult Lepidoptera are also included (Salvato 2001; Hoang et al. 2011). Sublethal reactions to insecticides include weight loss in caterpillars, changes in caterpillar development and pupation times, changes in chemical communication and mating behaviour of adult moths, and reduced reproduction of adult moths (e.g. Clark & Haynes 1992; Abro et al. 1993; Han et al. 2012; Pecenka & Lundgren 2015). Next to insecticides, herbicides may also have negative effects on Lepidoptera by reducing the occurrence, flowering or quality of certain host or food plant species (Schmitz et al. 2013; Hahn et al. 2014; Schmitz et al. 2014a,b).

Agricultural pesticide applications can result in lethal and sublethal effects on FVI species. This has been shown in laboratory studies mostly with honey bees but also other bee and some lepidopteran species. However, there is a need for further investigations of acute and chronic toxicity in wild bees, lepidopterans (especially moths) and species of other FVI groups with a focus on effects below lethal levels of pesticides. Effects of other pesticide classes than neonicotinoids, mixture toxicity and combined effects with other stressors such as parasites and pathogens should be investigated. Test species should be selected according to their ecological attributes as representatives of FVI (sub)groups. Furthermore, the impact of field-relevant pesticide doses should also be studied in ecologically more relevant field and semi-field experiments using other FVI organisms than the honey bee or bumble bee species. This research program should incorporate toxic effects of pesticide product adjuvants as well as the indirect effects of pesticides and their impact on ecosystem services of FVIs and their population responses. Moreover, field experiments should be designed to allow for at least some inference on source-sink dynamics. Otherwise, these effects could be simulated in landscape-scale models that need to be developed and validated.

Recommendations for FVI risk assessment concept

As a conservative approach it should be assumed that FVIs are potentially exposed towards pesticides on all fields and crop types, where pesticides are applied. Thus, in addition to the off-field assessment an in-field risk assessment should be obligatory for the authorisation of all pesticide uses. Crop specific scenarios for pesticide risk assessment of FVIs were developed for the respective crops types (i.e. attractive crops, unattractive crops, crops harvested before flowering, weeds in the field, off-field). The scenarios combine specific exposure scenarios with potential effects on the respective life stage of the FVI groups, bees and Lepidoptera. The developed matrices shall provide guidance for the selection of relevant scenarios for the risk assessment for the intended specific pesticide use.

Most FVIs are highly mobile and move between non target off-field areas and field areas. The assessment of potential risks at the local scale will not protectively address adverse effects of pesticides applied in-field on the off-field population. Thus, we propose that a local scale risk assessment as well as a landscape scale risk assessment should be conducted.

For local scale risk assessment a tiered approach is proposed. Based on the information on ecology and vulnerability currently available as well as the current available standardised test systems a basic toxicity data set should be required for lower tier risk assessment:

- 2 bee species (larvae and adult), i.e. two of the proposed focal bee species.
- 1 Lepidoptera species (larvae and adult), i.e. *Aglais io* or *Aglais urticae.*

For <u>all crops</u> chronic effects should be assessed as part of the first tier in order to detect the occurrence of delayed effects of acute exposure. In tier 1 risk should be assessed for all contact and oral exposure pathways described by relating expected exposure to toxicity.

In case an acceptable risk cannot be demonstrated in lower tiers the risk assessment might be refined in a higher tier risk assessment at the local scale. By conducting adequate higher tier studies the applicant has to demonstrate that there is no unacceptable risk for vulnerable FVI species. Suitable approaches might be conducting semi-field/field effect studies to refine substance specific aspects or conducting exposure studies to refine pesticide residues in environmental matrices (e.g. nectar/pollen, plant, soil).

Due to the high mobility of most vulnerable FVIs groups, effects on FVIs at the landscape scale should be considered. At landscape scale modelling approaches are needed to assess the risk with respect to spatio-temporal variation in pesticide dynamics and the interaction with spatial dynamics of mobile FVI. As landscape-scale population-level modelling is very complex and requires high effort, EFSA (2015) proposes a lower tier within landscape scale modelling by using so called look-up tables. Look-up tables shall provide the results of pre-run modelling scenarios, which can then be re-checked for the specific data of the applicant (e.g. environmental fate, GAP, intended use etc.). The look-up tables to be developed for NTAs as proposed by EFSA (2015) would need to be adapted with respect to the spatial dynamics of mobile FVI to meet FVI specific risk assessment requirements.

Landscape-scale population-level pollinator model feasibility

Landscape-scale population-level models (LSPLM) for ERA are not new. However, these are not common due to the difficulties of developing and maintaining the model systems needed to support these models. An existing system for modelling landscape-scale impacts of pesticides on pollinators is the BEEHAVE model (Becher et al. 2014). BEEHAVE model is specifically a honey bee single colony model and not suitable to model populations of individuals in a landscape. Therefore, BEEHAVE is not considered useful as a basis for constructing models of other pollinators.

Currently, the only models that satisfy all the population and landscape requirements are developed under ALMaSS (Topping et al. 2003), a C++ system of models designed to support simulation of terrestrial populations in managed environments. The feasibility to develop a wild pollinator model within ALMaSS (Topping et al. 2003) for the risk assessment for wild pollinators was evaluated in the present project.

The following aspects are a prerequisite for any pollinator species considered for modelling:

- well understood and researched, hence there is data to parameterise the models;
- their resources required to simulate their behaviour and ecology are predictable from mapping or other spatially explicit data;
- their dynamics are not closely linked to other species requiring a specific modelling of that species e.g. as would be the case if we attempted to model lynx and Canadian hare. Note that potentially this may be a problem if a pollinator species is closely linked in its population dynamics to a parasite (e.g. brood parasites and kleptoparasites in bees).

As a general rule, it is possible to model any species if we have some basic ideas of how they move, breed and die. However, for a model to be acceptable for risk assessment of pesticides there are some basic credibility criteria that need to be adhered to. One of these is that the data used to parameterise and design the model is scientifically supportable. Therefore, those species where there is little or no scientific literature support are not considered feasible for this approach. Development of models is considered feasible for the bee species *Bombus terrestris, Osmia bicornis* and the Lepidoptera species *Maniola jurtina.*

Risk mitigation measures

For the protection of FVI against the effects of pesticides, risk mitigation measures can be implemented. A total of 13 application-related or landscape-related measures were proposed and evaluated according to their efficacy to reduce pesticide exposure of FVIs, according to possible effects on FVIs and feasibility and acceptability by farmers. Measures can have the aim to reduce pesticide entries in off-field non target areas (e.g. in-field buffer strips, high vegetation, no spray zones, conservation fallows) and consequently to reduce the pesticide exposure of FVI in these areas. Moreover, there are also measures which (additionally) aim to reduce the exposure of FVI in-crop (e.g. reduction of pesticide application rate, pesticide application in the evening when diurnal FVIs are not active). Landscape-related measures (e.g. in-field buffer strips, high vegetation, extension of small field margins, management of off-field habitats) have (additionally) the potential to promote populations of FVI in the agricultural landscape, because the change in landscape design leads to an enhancement in habitat availability and/or quality for FVI. Thus, pesticide effects on FVI might be compensated/reduced by these landscape-related measures.

With regard to the efficiency to promote FVIs best investigated mitigation measures are wildflower in-field buffer strips and fallows. For both the efficiency to promote FVIs was scientifically demonstrated in available literature. Literature revealed that in-field buffer strips seed with wildflowers (or with nectar and pollen mixture) are most effective in increasing abundance and species richness of bees, butterflies and hoverflies. Conservation headlands, buffer strips with natural vegetation or grassed buffer strips seem to be partly less effective for bee and butterfly populations. In general, the number of forage flowers is important for the effectiveness of flower strips. Regional seed mixtures containing indigenous plant species should be used and should contain key plant species of the FVI species of concern to promote the presence of food specialized FVI species. In general, FVI abundance might increase with increasing buffer width, because of higher amounts of food and nesting possibilities.

For conservation fallows positive effects were shown in literature for bumble bees, wild bees, wasps, butterflies and moths. Similar to in-field buffer strips, the flower diversity as well as the flowering coverage were reported as important factors for FVI communities in terms of abundance and diversity. The impact of fallows created by natural regeneration is not comprehensively investigated. However, there are indications that due to a high degree of diversity of plant communities natural regenerated fallows might be an effective tool to protect FVIs. Perennial fallows are providing foraging habitats in early season, because of a wider range of resource availability. Thus, FVI groups - which are already active during the early season – benefit from the supply of flowers in spring.

In addition, efficiency could also be demonstrated for further risk mitigation measures (e.g. hedges, sowing of seed mixtures, no spray zones, extension of small field margins). However, there are many measures with further need of research. Particularly in case of application related measures (e.g. reduction of application rate, spray drift reducing techniques) there are no studies available investigating the effects on FVIs. In general, focus of available studies is the potential to reduce pesticide entries in off-field habitats or pesticide inputs in-crop. Based on the efficiency of these measures to reduce pesticide exposure, it can be assumed that also effects on FVIs are reduced. But there are also some landscape-related measures (i.e. creation of nesting possibilities, leaving deadwood in fruit orchards, maintenance of hedges) where the available database in literature is insufficient. However, efficiency to promote FVIs can be assumed based on ecological considerations.

The long-term persistence of FVI populations in agricultural landscapes depends on a balance between population sources (e.g. semi-natural habitats) and population sinks (e.g. agricultural fields, pesticide influenced off-field habitats). In general, the persistence of FVI populations is promoted by strengthen the sources and/or reducing population sinks.

Possibilities to strengthen population sources:

- Preservation of large off-field habitats (e.g. grasslands, forest, nature reserves) in agricultural landscapes.
- Management of existing off-field habitats to increase pollen resources.
- Achieving a heterogeneous landscape by implementing linear structured mitigation measures (e.g. wildflower in-field buffer strips) and areal structured mitigation measures (e.g. fallows).

Possibilities to reducing population sinks:

- Reduction of pesticide applications (e.g. rate, intervals, frequency);
- Use of spray drift reducing techniques;
- Timing of the application;
- Management of vegetation in sink habitats with respect to improve the quality of existing habitats and to create new habitats in order to provide resources for growth, maintenance and reproduction of FVI populations.

However, certain FVI groups show a reversed pattern of the normally expected source-sink relationship, i.e. (semi-)natural habitat acting as a source and fields acting as a sink. Hence, it is important to have sufficient information with respect to habitat requirements and life cycle of the species of concern, to adequately predict the dynamics and the consequences of risk mitigation measures.

Ecological focus areas

The EU common agricultural policy (CAP) was reformed in 2013 amongst others with the aim to strengthen a sustainable, ecological agriculture through a "Greening" component of direct payments. Amongst others, the "Greening" includes the designation of ecological focus areas (EFAs) with the aim to maintain biodiversity and natural resources. In general farmers with more than 15 ha arable land, are obliged to designate at least 5% of their arable land as EFA in order to obtain direct payments through the 1st pillar of the EU Common Agricultural Policy. The different types of EFAs were also evaluated with respect to their potential to reduce pesticide entries in adjacent off-field habitats and to promote abundance and diversity of FVI communities in agricultural landscapes based on available literature. The efficiency to promote FVIs was classified as scientifically demonstrated in case of five EFA types (i.e. land lying fallows, hedges or wooded strips, field margins, strips of eligible hectares along forest edges). For all other types the available database is insufficient, but efficiency to promote FVIs is assumed based on ecological considerations.

When implementing EFAs the question raises if there are any minimum requirements for promoting FVIs. Important aspects are the size of such an area, number of EFAs, connectivity of biotopes, or the life-time of biotopes. The size of an EFA is an important characteristic. In general, with increasing size of an area, boundary effects are reduced (e.g. boundary effects caused by pesticide drift entries). Furthermore, greater areas provide more foraging and nesting habitats. However, FVIs might already benefit from small widths of buffer strips. Available data reveal that wildflower/grassy and natural regenerated buffer strips adjacent to field crops with a minimum width of 6 m might be sufficient to support a variety of FVI species. In addition to annual in-field buffers the presence of perennial buffers could be considered to be important. For land lying fallow a minimum area of 0.6 ± 0.4 ha were derived based on available studies to support a variety of FVI species. Furthermore, FVI communities might benefit from a mixture of annual and perennial fallows present in agricultural landscapes.

The percentages of EFAs necessary to sufficiently compensate pesticide effects on FVI populations are difficult to determine based on current available data. Current available studies report that percentages of 3-7% (Cormont et al. (2016)) of natural habitats or 7.5 % (Holland et al. (2015)) of uncropped land were appropriate for promoting FVIs. A further study (Oppermann et al. 2016; Maus et al. 2017) showed that the implementation of 10% enhancement areas (such as flower strips, fallows) in a 50 ha study region promoted clearly FVI abundance and diversity. However, lower percentages were not tested in this study. Based on these few available data it is not clear, which percentages is sufficient for improving FVI communities. Moreover, besides the quantity, the quality of EFA (i.e. quality of food/foraging and nesting resources) seems to play an important role.

The main aim of EFAs is the permanent protection and promotion of biodiversity in agricultural landscapes which includes amongst others also the diversity of FVIs. Applied pesticides in agricultural landscapes are suspected as one factor for the decline of biodiversity (Marshall & Moonen 2002; Brittain et al. 2010; Balmer et al. 2013; BMEL 2013). Therefore, EFAs should not receive pesticide inputs and should be protected from pesticide entries. There are different options to reduce pesticide entries in EFAs:

- No spray zones around EFAs;
- Uncultivated buffers around EFAs (e.g. as implemented in Switzerland). To increase the acceptance by farmers funding for the creation/maintenance of these buffer areas should be provided in order to compensate for losses of agricultural land and consequently a loss of crop yield;
- The use of drift reducing nozzles or end nozzles adjacent to EFAs.

Opportunities for funding of risk management measures

In the context of the Common Agricultural Policy (CAP) of the European Union EU-funds are available to support farmers via direct payments (1st pillar) and via an environment-friendly and sustainable development of rural areas (2nd pillar). Part of the 1st pillar is the greening-premium which plays there the most important role for the funding opportunities of risk mitigation measures. The greening-premium includes the measures crop diversification, permanent grasslands and ecological focus areas. Some of the risk mitigation measures proposed for the protection of FVIs in the agricultural landscape might be funded in the context of ecological focus areas, e.g. land lying fallow or buffer strips. Moreover, there are further ecological focus areas which might benefit FVIs.

The 2nd pillar comprises specific programs for sustainable and environment-friendly farming and rural development. The main supporting instrument in implementing the EU priorities for the development of rural areas is the European Agricultural Rural Development Fund (EAFRD). Every EU member state receives an allocation of this fund by providing EAFRD support programs. In Germany, 13 RDPs are conducted on federal state level for the funding period 2014-2020. These programs mainly include voluntary environmental and climate measures related to agriculture, as well as measures to improve animal welfare and foster organic farming. The agro-environmental measures contained in the RDPs of

Rhineland-Palatinate (EULLE 2015) and Saxony-Anhalt (EPLR 2015) and the joint program of Lower Saxony and Bremen (PFEIL 2015) were exemplarily reviewed for possible funding opportunities for risk mitigation measures. The analysis revealed that measures such as in-field buffer strips and mowing rhythm of off-field habitats (grasslands) are funded in all three federal states. High vegetation or leaving deadwood in fruit orchards are funded in one and two federal states, respectively.

Finally, we identified knowledge gaps in all chapters and highlighted research opportunities to further deepen our understanding of pesticides effects on FVIs and improve the existing regulatory pesticide risk assessment.

Zusammenfassung

Einleitung

In Agrarlandschaften werden mit dem Ziel einer Ertragssteigerung Pestizide in Kulturpflanzen angewendet, um sogenannte Schadorganismen zu kontrollieren. Allerdings werden dabei zwangsweise auch Nicht-Zielarten gegenüber diesen Chemikalien exponiert. Durch verschiedene Aufnahmewege wie zum Beispiel Wurzelaufnahme und daran anschließende Translokation innerhalb der Pflanze oder direkte Überspritzung der Pflanze können die Blüten von Kultur- und Wildpflanzen mit Pestiziden kontaminiert werden. Blütenbesuchende Insekten (*Flower-visiting insects* (FVI)) stellen eine funktionelle Gruppe von Insekten dar, die deshalb auf Grund ihrer Mobilität und ihrer Fouragier-Aktivität sowohl auf behandelten Anbauflächen als auch auf benachbarten Flächen besonders gegenüber Pestiziden exponiert sind. Blütenbesuchende Insektenarten tragen nicht nur zur Biodiversität und zum ästhetischen Wert einer Agrarlandschaft bei, sondern sind darüber hinaus auch wichtige Bestäuber von Kultur- und Wildpflanzen. Zur Gruppe der FVIs, die viele wichtige Bestäuber umfasst, gehören neben Bienen auch Fliegen, Käfer, Nacht- und Tagschmetterlinge, Wespen und Ameisen (Rader et al. 2015). Die Bestäubung von Pflanzen stellt eine zentrale Ökosystemdienstleistung dar, da 35% der globalen landwirtschaftlichen Produktion mit Kulturpflanzen verknüpft ist, deren Ertrag sich erhöht, wenn sie von Tieren bestäubt werden (Klein et al. 2007). Der globale ökonomische Wert der Tierbestäubung wurde auf 153 Milliarden Euro geschätzt (Gallai et al. 2009). Über ihre ökonomische Bedeutung hinaus ist biotische Bestäubung essentiell für den Erhalt der heimischen Flora, da 85% aller Blütenpflanzen von Tieren bestäubt werden (Ollerton et al. 2011). Blühende Kultur- und Wildpflanzen werden zumeist von Insekten bestäubt (Klein et al. 2007; Ollerton et al. 2011).

Aufgrund ihrer Bedeutung werden blütenbesuchende Insekten durch verschiedene Richtlinien, Deklarationen und Vorschriften sowohl auf europäischer Ebene als auch weltweit geschützt. Am 13. Treffen der Unterzeichner der Convention on Biological Diversity (CBD) erfolgte die Verabschiedung der "*Cancun Declaration on Mainstreaming the Conservation and Sustainable Use of Biodiversity for Well-Being*" (United Nations 2016). Die Mitgliedsstaaten verpflichten sich darin zu effektiven Maßnahmen um dem Verlust der Biodiversität entgegenzuwirken. Eine Handlungsempfehlung zur Erreichung dieser Ziele für den landwirtschaftlichen Sektor schließt das erfolgreiche Management und den Schutz von Bestäubern ein. Außerdem wurde von 12 EU Mitgliedsstaaten eine „*Coalition of the Willing on Pollinators*" gegründet, die sich dem Schutz der Bestäuber verpflichtet (Coalition of the Willing on Pollinators 2016).

Es zeichnet sich zunehmend ab, dass FVI-Bestände weltweit abnehmen. Als Gründe für diese Entwicklung werden Faktoren wie Klimawandel, Habitatverlust und -fragmentation, Umweltverschmutzung und Pestizideinsatz diskutiert (Goulson et al. 2015). Diese Entwicklung ist besonders bei Bienenarten deutlich sichtbar. In Deutschland stehen 53% der Wildbienenarten auf der Roten Liste (Westrich et al. 2011), in einigen Europäischen Ländern bis zu 65% (Patiny et al. 2009). Seit dem letzten Jahrhundert werden in der USA und Europa dramatische Abnahmen der Anzahl der Honigbienen (*Apis mellifera*) Kolonien sowie eine Verringerung der Wildbienen-Diversität beobachtet (Natural Research Council 2006; vanEngelsdorp et al. 2008; Potts et al. 2010; Goulson et al. 2015; Potts et al. 2015). Darüber hinaus wurden Populationsabnahmen von Schmetterlingen, Motten und Schwebfliegen in der EU beobachtet (EASAC 2015; Gilburn et al. 2015; Potts et al. 2015). Pestizideffekte auf blütenbesuchende Insekten wurden vor allem für Bienen festgestellt. Im Fokus der Untersuchungen standen hierbei in der Vergangenheit insbesondere die Honigbienen, da sie wichtige Bestäuber darstellen und damit ein wirtschaftliches Interesse besteht, überlebensfähige Populationen zu erhalten (Klein et al. 2007; Gallai et al. 2009). Allerdings gibt es neben Bienen viele andere Gruppen blütenbesuchender Insekten, die gegenüber Pestiziden in der Agrarlandschaft exponiert und daher betroffen sind (Godfray et al. 2014; Gilburn et al. 2015; Godfray et al. 2015). Daher sollten die Begriffe

„Bestäuber“ und „blütenbesuchende Insekten“ unterschieden werden und die gesamte Gemeinschaft der FVI betrachtet werden, wenn der Einfluss von in der Landwirtschaft eingesetzter Pestizide untersucht wird. Im Kontext dieses Berichts bezieht sich der Begriff blütenbesuchende Insekten (FVI) auf Insektengruppen, die zumindest in Laufe eines Lebensstadiums Blütenressourcen wie Nektar und Pollen nutzen.

Regulatorische Entwicklung

Die derzeit gültige Bewertungsleitlinie zur Terrestrischen Ökotoxikologie unter der Richtlinie des EU Rats 91/414/EEC (SANCO 2002) bezieht sich für die Risikobewertung für Bienen auf ein Dokument der Europäischen und Mediterranen Pflanzenschutz Organisation (OEEP/EPPO 2010a, b). Hierbei wird die Honigbiene als einzige Stellvertreterart für alle Bienen genutzt. Auf Grund der großen ökologischen Unterschiede zu den meisten anderen Arten wird dieses Vorgehen jedoch kritisch diskutiert (Arena & Sgolastra 2014; Rundlöf et al. 2015; Stoner 2016). Darüber hinaus wurden weitere Defizite am aktuellen Vorgehen identifiziert wie zum Beispiel die Aufnahme relevanter Expositionsszenarien oder die statistische Aussagekraft von Freiland und Halbfreiland Studiendesigns (EFSA PPR Panel 2012; EFSA 2013). Das Risiko für alle nicht-Bienen FVIs soll derzeit durch die Nicht-Ziel Arthropoden (NTA, *Non-target Arthropods*) Risikobewertung abgedeckt werden, die sich auf Ergebnisse des ESCORT2 Arbeitstreffens beziehen (Candolfi et al. 2001). Auch dieser Ansatz weist klare Defizite auf, da weder blütenbesuchende Insekten in der Risikobewertung berücksichtigt werden, noch die orale Aufnahme von Pestiziden in den dazugehörigen Studien untersucht wird.

Die Europäische Behörde für Lebensmittelsicherheit (EFSA) erkannte diese Defizite und entwarf eine neue Bewertungsleitlinie für Bienen (EFSA 2013) sowie ein wissenschaftliches Gutachten zu NTAs (EFSA 2015), die substantielle Verbesserungen zum Risikobewertungsprozess von FVIs enthalten. Allerdings erschwert die Tatsache, dass Bienen und alle anderen FVIs verschiedenen Risikobewertungsschemen zugeordnet werden, die Implementierung eines effizienten Risikobewertungsprozesses für die gesamte Gruppe der FVIs. Ein weiteres wichtiges Problem ist die Auswahl von Testarten und deren Repräsentativität. Die Repräsentativität der Honigbiene für die gesamte Lebensgemeinschaft der Bienen (Honig- und Wildbienen), geschweige denn andere FVI-Gruppen, ist aufgrund ihrer physiologischen und ökologischen Eigenschaften fraglich (Cutler et al. 2014; Rundlöf et al. 2015). Die Vorauswahl der zusätzlichen Testspezies *Bombus terrestris* und *Osmia bicornis* ist ebenfalls strittig, da Unsicherheiten sowohl hinsichtlich ihrer Eignung als Modellorganismen für die Risikobewertung als auch hinsichtlich ihrer Empfindlichkeit als Testorganismen bestehen. Aufgrund der ökologischen Unterschiede, insbesondere in Bezug auf ihre Sozialität, könnten die beiden vorgeschlagenen zusätzlichen Testarten jedoch geeignet sein, die Risikobewertung auf höherer Ebene zu verbessern (Cutler et al. 2014; Rundlöf et al. 2015). Die Einführung einer Schmetterlings-Surrogat-Testspezies für Nicht-Bienen-FVIs ist eine vernünftige Maßnahme, aber diese zusätzliche Testspezies sollte mit großer Sorgfalt ausgewählt werden. Es ist unklar, ob eine Spezies ausreicht, um diese ökologisch vielfältige Gruppe zu repräsentieren, da nur wenige Informationen über die Empfindlichkeit von Nicht-Bienen-FVI-Arten vorliegen. Darüber hinaus sind Expositionsszenarien für eine so heterogene Gruppe wie FVIs schwer zu definieren, da es bisher keine umfassende Überprüfung des Expositionsrisikos in allen FVI-Taxa gibt. Diese Expositionspfade sollten auch bei der Bewertung der Effekte berücksichtigt werden (z. B. werden orale Pestizideffekte auf herbivore FVI-Larvenstadien nicht berücksichtigt). Darüber hinaus müssen adäquate und durchführbare Labor- und (Halb-) Freilandtestprotokolle für die vorgeschlagenen zusätzlichen Testspezies entwickelt und standardisiert werden.

Insgesamt berücksichtigen die jüngsten EFSA-Revisionen der Bienen (EFSA-PPR-Panel 2012; EFSA 2013) und NTA-Risikobewertung (EFSA 2015) noch immer nicht ausreichend die spezifischen ökologischen Eigenschaften der verschiedenen FVI-Taxa hinsichtlich Exposition und potentieller Effekte. Für eine protektive Risikobewertung sollten zudem komplexere Probleme wie die indirekten Auswirkungen von Herbiziden auf FVIs durch Nahrungsverringerung, landschaftsskalige Quellen-

Senken-Effekte oder Auswirkungen auf FVI-Ökosystemdienstleistungen (z.B. Bestäubung, Biodiversität) in die Bewertung einfließen.

Taxonomische Gruppen von blütenbesuchenden Insekten

In der wissenschaftlichen Literatur werden Bienen, Schwebfliegen und Lepidopteren oft als FVI-Hauptgruppen genannt (Winfree et al. 2011). Jüngere Studien belegen jedoch eine deutlich höhere Diversität von FVI-Gemeinschaften in der Agrarlandschaft, so dass die derzeitige Definition relevanter Gruppen unvollständig erscheint. Grass et al. (2016) zeigten, dass Blüten in Wildblumenanpflanzungen in der mitteldeutschen Agrarlandschaft neben Bienen und Schwebfliegen von einer vielfältigen Gemeinschaft anderer Insektenarten besucht wurden. Tatsächlich repräsentierten Nicht-Bienen und Nicht-Schwebfliegen-Insekten die Hälfte der besuchenden Individuen und 75% der FVI-Arten. Darüber hinaus waren Nicht-Schwebfliegen-Dipteren der weitaus größte Teil der Insektenarten. Im Gegensatz dazu hatten Schmetterlinge nur einen geringen Anteil an der FVI-Abundanz, während die Blütenbesuche von Bienen und Nicht-Schwebfliegen-Dipteren in Bezug auf Anzahl der Individuen mit denen von Honigbienen vergleichbar waren. Weitere Metaanalysen sowohl in Nutzpflanzen als auch in Nicht-Zielgebieten liefern zusätzliche Hinweise darauf, dass FVI-Gemeinschaften diverser sind als bisher angenommen (Orford et al. 2015; Rader et al. 2015).

Anhand aktueller Daten können Bienen, Fliegen (Nicht-Syrphiden und Syrphiden), Motten und Schmetterlinge sowie Käfer als relevante FVI-Gruppen in Nutzpflanzen und deren naturnaher Umgebung definiert werden. Die prozentualen Anteile der jeweiligen Gruppen an der Gesamtzusammensetzung der FVI-Gemeinschaften in Bezug auf Arten- und Individuenzahlen sind in verschiedenen Kultursystemen (Rader et al. 2015) und in naturnahen Lebensräumen (Grass et al. 2016) stark variable. Für Nicht-Bienen-Hymenopteren und Hemiptera stehen nicht genügend Informationen zur Verfügung, um eine Aussagen über ihre Relevanz als FVIs treffen zu können. Trotz eindeutiger Identifizierung einer Vielzahl bedeutender FVI-Gruppen sind ausreichende Informationen zur Bewertung des Pestizidrisikos nur für Bienen und Schmetterlinge verfügbar. Dagegen bestehen große Wissenslücken bezüglich der ökologischen Information für Fliegen und Käfer. Daher konzentriert sich dieser Bericht hauptsächlich auf Bienen und Schmetterlinge.

Habitate von blütenbesuchenden Insekten

Die Agrarlandschaft bietet vielfältige Lebensräume für FVIs, die in drei allgemeine Typen unterteilt werden können: Kulturpflanzen (*in-crop*/*in-field*), gemanagte Blühstreifen (*off-crop*/*in-field*) und naturnahe Feldrandstrukturen (*off-crop*/*off-field*). Diese Bereiche unterscheiden sich in vielen Aspekten, wie struktureller Zusammensetzung, Arteninventar/Diversität der Pflanzengemeinschaft, Verfügbarkeit von Nahrungsressourcen in Raum und Zeit, Abundanz und Diversität natürlicher Räuber oder anthropogenem Stress wie Pestizideinsatz. Anbauflächen unterscheiden sich in ihrer Eignung als Habitate für Bienen aufgrund der Attraktivität der Kulturpflanzen. Bei anderen FVI-Arten ist es jedoch aufgrund der knappen Datenlage ungleich schwieriger, die Attraktivität von Kulturpflanzen als Habitate zu bewerten. Grundsätzlich können aber für FVI-Gruppen wie Lepidoptera oder Käfer Nutzpflanzen nicht nur für adulte Stadien, sondern auch für ihre herbivore Larvenstadien einen Lebensraum darstellen. Darüber hinaus können selbst nicht attraktive Anbauflächen Habitate für FVIs bieten, wenn z. B. ein Unterwuchs mit Wildpflanzen im Feld besteht (Storkey & Westbury 2007; Balmer et al. 2014; Manandhar & Wright 2016). Sogar wenn auf landwirtschaftlichen Anbauflächen keine Blütenpflanzen vorhanden sind, können diese Flächen Lebensraumfunktionen für FVIs bereitstellen (z. B. Nistbereich für Bienen, Fliegen und Käfer oder temporäre Zuflucht für alle fliegenden FVI-Lebensstadien). In der Agrarlandschaft gibt es neben den Anbauflächen verschiedene Arten von feldnahen Strukturen, die sich hinsichtlich des Pflanzenbestands und ihrer Eignung als Habitat für FVIs unterscheiden. In intensiv bewirtschafteten landwirtschaftlichen Gebieten stellen diese feldnahen Strukturelemente die wichtigste Art halbnatürlicher Lebensräume dar und bieten eine Mehrfachlebensraumfunktion für FVI-Arten (Marshall & Moonen 2002). Zum Beispiel sollen

Blühstreifen als Agrarumweltmaßnahmen geeignete Lebensräume für FVI-Arten schaffen (z. B. Garibaldi et al. 2014; Feltham et al. 2015).

Innerhalb der Agrarlandschaft sind Kulturpflanzen sowie Feldrandstrukturen und Blühstreifen Lebensräume von FVI-Arten. FVI-Arten nutzen im Laufe ihres Lebenszyklus verschiedene von den unterschiedlichen Habitaten angebotene Ressourcen. In Abhängigkeit von den ökologischen Eigenschaften der FVI-Arten können die genutzten Ressourcen während ihres Lebenszyklus zeitlich und räumlich variieren (z. B. Luftraum, Blüten, Stängel/Blätter, Boden, Wasserreservoire).

Ökologische Kategorien und Vulnerabilität

Die spezifischen ökologischen Merkmale einer Spezies sind ausschlaggebend für die Vulnerabilität von FVI-Populationen gegenüber Umweltveränderungen (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015). Daher ist es möglich, FVI-Arten anhand von ökologischen Merkmalsdaten in ökologische Kategorien einzuordnen und ihre Vulnerabilität gegenüber Stressoren wie Pestiziden zu bewerten.

Bienen (Hymenoptera: Apoidea: Anthophila) sind eine monophyletische Gruppe, die in Europa mehr als 1900 Arten umfasst (Nieto et al. 2014). Neben der bekannten westlichen Honigbiene *Apis mellifera* gibt es in Europa eine Vielzahl ökologisch variabler Wildbienenarten. Einige Arten sind zum Beispiel eusozial, d. h. Leben in Kolonien oder Ansammlungen. Jedoch leben die meisten Bienenarten solitär. Zusätzlich gibt es viele parasitäre Arten, die ihren Wirt parasitieren, um ihre Nachkommen zu ernähren (Michener 2007; Goulson 2010). Es gibt verschiedene Niststrategien bei Bienen: Die meisten Arten graben sich in den Boden, um ihr Nest zu bauen, andere besetzen bereits vorhandene Hohlräume im Boden oder Totholz oder bauen Nester aus gesammeltem Material (Michener 2007).

Auf Basis der Daten einer umfassenden Datenbank für alle europäischen Bienenarten (Roberts et al. unveröffentlicht) wurden wichtige ökologische Merkmale (Mobilität, Sozialität, Nestart, Lektie, Flugzeit/-dauer, Voltinismus) analysiert, um europäische Bienenspezies in ökologische Vulnerabilitätsklassen einzuteilen, sowie für diese Klassen repräsentative Arten auszuwählen. Die europäischen Bienenarten konnten unter Berücksichtigung dreier ökologischer Merkmale (Mobilität, Lektie, Sozialität) in sieben ökologische Kategorien eingeteilt werden. Anschließend wurden repräsentative Stellvertreterarten in den jeweiligen Kategorien identifiziert, wobei eine weite Verbreitung in Europa sowie repräsentative Merkmalskombinationen unter besonderer Berücksichtigung einer repräsentativen Flugaktivität während des ganzen Jahres berücksichtigt wurden. Darüber hinaus wurden die drei Hauptmerkmale, die für die Kategorisierung von Bienenarten identifiziert wurden, plus die Flugsaison/-länge für die qualitative Bewertung der ökologischen Vulnerabilität verwendet. Von den sieben Kategorien wurden drei als hoch vulnerable, drei als mittel vulnerable und nur eine als wenig vulnerable eingestuft. Diese Ableitung von Gruppen unterschiedlicher Vulnerabilität sollte durch aktuelle Populationsdaten aus mittel- bis langfristigen ökologischen Studien bestätigt werden. Leider sind solche Daten für die meisten europäischen Bienenarten selten. Solche Daten sollten in einem europaweiten Bienen-Populations-Monitoring erhoben werden.

Motten und Schmetterlinge (Lepidoptera) sind eine weit verbreitete und artenreiche phytophage Insektengruppe. Obwohl Lepidoptera eine der am meist untersuchten Arthropodengruppen darstellen, konzentrierte sich in der Vergangenheit Forschung zu Lepidoptera auf tagaktive Schmetterlinge (New 2004), die jedoch nur 10% der Lepidoptera-Arten repräsentieren (Shields 1989). Die restlichen Arten werden als Motten oder Nachtfalter klassifiziert, die oft eine dämmerungs- und nachtaktive Lebensweise aufweisen. Die juvenilen Lebensstadien (Larven, Raupen) der meisten Lepidoptera-Arten sind Pflanzenfresser und ernähren sich von Pflanzenmaterial wie Blättern, Wurzeln, Blüten, Samen oder Früchten (Scoble 1995). Da sich einige Lepidoptera-Arten während ihres Raupenstadiums auch von Feldfrüchten ernähren, werden diese Arten als landwirtschaftliche Schädlinge eingestuft. Die Mehrheit der Motten- und Schmetterlingslarven ernährt sich jedoch von

Nicht-Kulturpflanzen (Ebert 1994; Scoble 1995). Im adulten Stadium ernähren sich die meisten Arten von Nektar und bisweilen von Pollen (Scoble 1995).

Verschiedene ökologische Merkmale von Lepidopteren korrelieren mit der Vulnerabilität von Lepidoptera-Arten (insbesondere von Schmetterlingen) gegenüber Umwelt- und Klimaveränderungen (Aguirre-Gutierrez et al. 2016; Eskildsen et al. 2015). Zu diesen Merkmalen gehören die Mobilität adulter Stadien, die Habitat- und Wirtspflanzenspezialisierung der Raupen sowie das Überwinterungsstadium. Darüber hinaus könnten weitere Merkmale, wie die Dauer der Flugperiode der adulten Stadien oder die Wachstumsbedingungen von Wirtspflanzen hinsichtlich des Stickstoffeintrags (Eutrophierung) von Bedeutung sein (Franzen & Johannesson 2007; Kuussaari et al. 2007; Barbaro & van Halder 2009; Aguirre-Gutierrez et al. 2016). Im Gegensatz zu Bienen ist eine Zuordnung von Lepidoptera-Arten zu ökologischen Kategorien sowie eine Bewertung dieser Kategorien hinsichtlich ihrer Vulnerabilität derzeit jedoch nicht möglich, da für europäische Nachtfalter- oder Schmetterlingsarten keine umfassende Datenbank ökologischer Merkmalsdaten zur Verfügung steht. Aufgrund des Mangels an Daten ist auch eine Auswahl von Stellvertreterarten für die Risikobewertung von Pestiziden schwierig. Jedoch bieten sich mehrere Lepidopteren Arten als ökotoxikologische Testspezies an, zum einem aufgrund ihrer weiten Verbreitung und zum anderen aufgrund der Tatsache, dass sie bereits in ökotoxikologischen Studien verwendet wurden. Zur Ableitung von Stellvertreterarten für die Risikobewertung von Motten, Schmetterlingen und anderen Nicht-Bienen-FVI sind weitere Untersuchungen zu deren ökologischen Eigenschaften notwendig.

Exposition von Lebensräumen

Pestizide werden über verschiedene Eintragspfade in FVI-Lebensräume eingetragen. Diese können im Allgemeinen zwei verschiedenen Gruppen zugeordnet werden: Primäre Prozesse charakterisieren die beabsichtigte Anwendung von Pestiziden auf einer Nutzpflanze (*in-field*). Diese Methoden umfassen die Sprühapplikation von meist nicht-systemischen Pestiziden sowie die Anwendung als feste Formulierung (Saatgutbehandlung, Granulat), Stängelinjektion oder Bewässerung mit systemischen Pestiziden. Darüber hinaus wird ein Teil, der auf Agrarflächen angewendeten Pestizide, unbeabsichtigt durch sekundäre Prozesse wie zum Beispiel Abdrift, Feldrand-Overspray, Staubdispersion/-drift – sowie Run-off in angrenzende Habitate eingetragen. Durch diese primären und sekundären Prozesse können Pestizide in die verschiedenen für FVI relevanten Habitats- und Umweltkompartimente eingetragen werden (z.B. Luftraum, Pollen und Nektar von Blüten, Stängeln und Blättern von Pflanzen, Guttationswasser, extraflorale Nektarien, Honigtau, kleine ephemere Wasserkörper (z.B. Pfützen), größere Gewässer (Flüsse, Seen) und Boden in Feld- und *off-field*-Lebensräumen).

Exposition von blütenbesuchenden Insekten

Der Eintrag von Pestiziden in FVI-Habitate führt nicht notwendigerweise zur Exposition von FVI-Individuen bzw. Populationen. Jedoch können zur Identifikation relevanter Expositionspfade und zur Abschätzung der Aufnahmewahrscheinlichkeit über die verschiedenen Habitatmatrizes Informationen über ökologische Merkmale (*life history traits*) der FVI-Arten verwendet werden. Allerdings sind umfassende Trait-Informationen nur für Bienen verfügbar, deshalb ist eine detaillierte Analyse anderer FVI-Gruppen aus Mangel an Daten nicht möglich. Für Bienen konnten mehrere Merkmale identifiziert werden, die das Ausmaß ihrer Exposition maßgeblich beeinflussen:

- Die jährliche Flugperiode der meisten Wildbienenarten fällt mit der Hauptanbauzeit und somit mit dem Zeitfenster der meisten Pestizidanwendungen zusammen. Daher kann grundsätzlich davon ausgegangen werden, dass abhängig von der Kultur Bienen gegenüber einer Vielzahl von Pestiziden exponiert werden könnten. Darüber hinaus besteht die Möglichkeit einer potentielle Exposition gegenüber Pestiziden, die außerhalb der aktiven Flugperiode angewendet werden durch Aufnahme von langlebigen Verbindungen in Boden und Pflanzenmaterial (Fantke & Juraske 2013; Goulson 2013b; Bonmatin et al. 2015).

- Abgesehen von ihrem jährlichen Aktivitätsfenster unterscheiden sich die verschiedenen Bienentaxa auch in ihren tageszeitlichen Aktivitätsmustern. Daher könnten Maßnahmen zur Reduktion der Exposition der Honigbiene, wie die Verzögerung der Anwendung von Pestiziden in die Abendstunden, zum Anstieg der Exposition anderer Bienenspezies (z.B. Hummeln) führen.
- Unterschiedliche Strategien in der Nahrungswahl können die Exposition von Bienen beeinflussen. Z.B. können polylektische Bienenarten gegenüber Pestizidrückständen aus einem breiten Blütenspektrum exponiert sein, während das Expositionsmuster oligolektischer Bienenarten aufgrund ihres schmalen Nahrungspflanzenspektrums auf die Rückstände aus Blüten von wenigen Pflanzenarten zurückzuführen ist.
- Die Art der Nistweise der Bienen ist ein relevanter Faktor bei der Beurteilung ihres Expositionspotentials. Da sich die Mehrzahl der Bienenarten (64%) in den Boden graben, um ihre Nester zu bauen ("*ground excavator*"), sollte die Exposition über den Boden als kritischer Pfad angesehen werden. Darüber hinaus haben weitere artspezifische Merkmale der Nistweise einen Einfluss auf die potentielle Exposition. Zum Beispiel sammeln einige Bienenarten Pflanzenmaterial aus verschiedenen Quellen (z.B. von Wildpflanzen im *in-crop* und/oder *off-crop* Bereich) für den Bau ihrer Nester und können so gegenüber Rückständen im gesammelten Pflanzenmaterial exponiert sein. (Westrich 1990).
- Die Wahrscheinlichkeit einer Exposition sozialer Bienen gegenüber Pestiziden ist aufgrund der bloßen Anzahl von Nahrungssammlern und der Menge an gesammeltem Pollen aus einer Vielzahl von Pflanzen im Vergleich zu solitären Bienen höher (Brittain & Potts 2011).

Die Bewertung expositionsrelevanter Merkmale für Lepidoptera ist ungleich schwieriger als für Bienen, da hierfür keine umfassende Datenbasis mit Trait-Informationen zur Verfügung steht. Daher können zwar ökologische Attribute von Lepidoptera-Arten aufgelistet werden, aber ihre Relevanz hinsichtlich ihres Einflusses auf die Exposition von Schmetterlings- und Mottengemeinschaften bleibt offen und muss in Folgestudien evaluiert werden.

Untersuchungen von Pestizidrückständen in FVI-Individuen sind unerlässlicher Bestandteil einer quantitativen Bewertung der Pestizidbelastung. Derzeit sind solche Daten allerdings nur für Bienen verfügbar. Um eine angemessene Expositionsbewertung für andere FVI-Gruppen zu ermöglichen, sollten Daten auch für nicht-Bienen FVI-Taxa erhoben werden (z.B. Schmetterlinge und Motten, Fliegen oder Käfer). Für Bienen ist die Exposition gegenüber einer Vielzahl von Pestiziden nachgewiesen: Zum Beispiel wurden in Studien, die Pestizidrückstände in Honigbienenbrut, Pollen und Honig analysierten, bis zu 98 aktive Substanzen und Abbauprodukte gefunden (Chauzat 2006; Mullin et al. 2010). In einem Literatur-Review über die Umweltauswirkungen von Neonikotinoiden stellten Wood & Goulson (2017) zu erwartende Rückstände (Berechnungen der EFSA auf Basis von Freiland-Feldstudien) von drei Neonikotinoiden in Pollen und Nektar für ausgewählte Nutzpflanzen zusammen. Zwar schwankte die Höhe der Rückstände zwischen den verschiedenen Nutzpflanzen, aber die Dosen in Pollen waren konsistent höher als in Nektar.

Bienen und andere FVI-Gruppen sammeln Pollen und Nektar von einer Vielzahl von Pflanzen auch in *off-crop* Lebensräumen. Die Vegetation in diesen Nicht-Zielgebieten (z.B. Feldränder neben Feldern) ist oft ebenfalls mit Pestiziden kontaminiert (Wood & Goulson 2017). Zum Beispiel konnten Long & Krupke (2016) zeigen, dass Wildbienen gegenüber einer Vielzahl von Pestiziden exponiert sind, wenn sie in Nicht-Zielgebieten auf Nahrungssuche gehen. Neuere Studien berichten über mittlere Konzentrationen von Neonikotinoiden in Pollen und Nektar von Wildpflanzen von < 0.4 bis 14.8 ng/g bzw. < 0.1 bis 1.5 ng/g. Diese Rückstandsmengen sind vergleichbar mit Rückständen in saatgutbehandelten Kulturpflanzen während der Blüte (Wood & Goulson 2017). Um die Kontamination von Pollen mit der Exposition von Bienen zu verknüpfen, ist es notwendig, von Bienen gesammelte Pollenquellen zu untersuchen. In Studien mit freifliegenden Honigbienen als Pollensammler zeichnen sich erkennbare Trends ab: Die höchsten Rückstandsmengen finden sich, wenn ein großer Anteil Pollen von Kulturpflanzen gesammelt wird. Wenn umgekehrt die Anteile von

Wildblumen im gesammelten Pollen höher sind, sind die Neonikotinoid-Rückstände niedriger (Wood & Goulson 2017). Im Allgemeinen kann davon ausgegangen werden, dass eine akute Exposition durch hohe Dosen in gesammelten Pollen und Nektar zeitlich mit der Blüte von FVI-attraktiven Pflanzen zusammenfallen. Die chronische Exposition von Arten mit einer langen aktiven Flugperiode wie Honigbienen könnte jedoch durch die Nahrungssuche auf Wildpflanzen bestimmt werden (Botías et al. 2015; Wood & Goulson 2017).

Neben der Exposition über direkten Overspray oder Sprüh-/Staubabdrift sowie der Aufnahme von Pestizidrückständen aus Pollen und Nektar sollten weitere Expositionspfade als relevant für blütenbesuchende Insekten betrachtet werden:

- Die Mehrheit der europäischen Bienenarten nistet im Boden, daher kann die Pestizidbelastung durch Bodenkontakt einen relevanten Expositionspfad darstellen. Eine Exposition über Boden kann auch für die bodenlebende Lebensstadien anderer FVI-Gruppen (z. B. Fliegen- und Käferlarven) relevant sein. In mehreren Studien konnten z.B. Rückstände der Neonikotinoide Imidacloprid, Clothianidin oder Thiamethoxam in landwirtschaftlichen Böden im Bereich von 0.4 bis 13.3 ng/g nachgewiesen werden. Für die Bewertung der Exposition von FVIs über den Boden ist neben den Spitzenkonzentrationen der jeweiligen Substanzen auch deren Persistenz in der Bodenmatrix zu berücksichtigen, die z.B. für Neonikotinoide zwischen einigen Tagen und Jahren variiert (Goulson 2013a). Es ist jedoch schwierig, die Kontamination von Böden mit Pestiziden mit der FVI-Exposition durch Bodenkontakt zu verknüpfen.
- FVI-Spezies können gegenüber Pestizidrückständen in oder auf Pflanzenmaterial exponiert sein (z.B. pflanzenfressende Lebensstadien von Lepidopteren und Käfern oder Nestmaterialien sammelnde Bienen). Relevante Rückstände sind in Stängeln oder –blättern von Wildpflanzen in vergleichbarer Höhe wie in Nutzpflanzen nachweisbar (Botías et al. 2016; Mogren & Lundgren 2016).
- Blütenbesuchende Insekten können gegenüber toxikologisch relevanten Pestizidkonzentrationen exponiert werden, wenn sie Wasser aus kleinen ephemeren Wasserkörpern (z. B. Pfützen) auf Agrarflächen konsumieren (Samson-Robert 2014; Schaafsma et al. 2015).
- Im Guttationswasser, das von einigen Pflanzen abgesondert wird, konnten Pestizidkonzentrationen nachgewiesen werden, die die Konzentrationen in Nektar um vier bis fünf Größenordnungen übersteigen (Godfray et al. 2014). Eine Exposition von FVI-Individuen (z.B. Bienen) durch die Konsumtion von Guttationswasser kann nicht ausgeschlossen werden. Allerdings bleibt ein klarer Nachweis der Aufnahme aus Pestizidrückständen in Guttationstropfen für z.B. Bienen jedoch offen (Wood & Goulson 2017).

Um die Exposition von FVI-Arten über die Vielzahl relevanter Pfade in der Risikobewertung für Pestizide zu adressieren, wurden in jüngster Vergangenheit auf europäischer Ebene diverse Konzepte zur Expositionsabschätzung vorgeschlagen (EFSA 2013, 2015; SANCO 2014). Die vorgeschlagenen Schätzverfahren konnten im Rahmen der vorliegenden Arbeit in einigen Fällen auf Basis von Ergebnissen aus wissenschaftlichen Studien angepasst und verbessert werden. In anderen Fällen sind allerdings derzeit keine adäquaten theoretischen Konzepte zur Expositionsabschätzung verfügbar. In diesen Fällen sollte im Rahmen der Risikobewertung eine chemische Rückstandsanalyse relevanter Matrizes angewendet werden.

Um die quantitative Bewertung der FVI-Exposition zu verfeinern sowie adäquate Expositionsmodelle zu erstellen und diese zu validieren, sollte die dafür als Grundlage dringend notwendige Datenbasis verbreitert werden. Hierzu sollten:

- Nahrungsaufnahmeraten (z.B. für Nektar- und Pollen) für die relevanten Lebensstadien von nicht-Bienen FVI-Gruppen (z. B. Lepidoptera, Fliegen und Käfer) gesammelt werden (insbesondere in *off-field*-Habitaten);

- Daten zu Pestizidrückständen in allen relevanten Matrices gesammelt werden (insbesondere in *off-field*-Nichtzielpflanzen).

Da FVIs mobile Spezies sind, ist für eine protektive Bewertung möglicher Risiken für FVI-Populationen das Wissen über ihr räumlich-zeitliches Expositionsmuster erforderlich. Dies wurde auch von der EFSA in ihrem NTA-Gutachten (EFSA 2015) anerkannt. Um die FVI-Exposition in Raum und Zeit zu bewerten, können Modellierungsansätze auf Landschaftsmaßstab ein wertvolles Werkzeug darstellen.

Pestizidwirkung auf blütenbesuchende Insekten

Da in den Datenanforderungen für die Risikobewertung im Rahmen des EU-Zulassungsverfahrens von Pflanzenschutzmitteln Daten zur Toxizität auf Honigbienen gefordert werden, sind für diese Spezies Daten zur akuten Toxizität für alle registrierten Pestizide verfügbar. Die Empfindlichkeit anderer Bienenarten ist jedoch in der Regel unbekannt, so dass es schwierig ist, die Honigbiene als Stellvertreterorganismus für die Risikobewertung von Wildbienen oder anderen FVIs zu etablieren. Arena & Sgolastra (2014) analysierten die verfügbare Literatur und schlugen einen sogenannten *Bridging-Faktor* von 10 zum LD50 der Honigbiene vor, um speziesspezifische Unterschiede in der Sensitivität zwischen Honigbienen und Wildbienen in 95% aller Fälle abzudecken. Uhl et al. (2016) bewerteten die akute Toxizität von Dimethoat gegenüber mehreren europäischen Wildbienenarten und kamen zu dem Schluss, dass bei Anwendung des SSD-Konzepts dieser *Bridging-Faktor* eine protektive Bewertung der mit Dimethoat verbundenen Risiken ermöglichen würde. Die Autoren stellten jedoch auch fest, dass die relative Empfindlichkeit für verschiedene Pestizide unterschiedlich ist und dass der *Bridging-Faktor* möglicherweise für andere Pestizide angepasst werden muss. Darüber hinaus gibt es eine große Varianz beim Vergleich der akuten Sensitivität von Wildbienenarten und der Honigbiene gegenüber verschiedenen Pestiziden (Uhl et al. in prep.). Folglich besteht zur Ableitung eines wissenschaftlich fundierten Sicherheitsfaktors, der die Unsicherheiten aufgrund speziesabhängiger Sensitivitätsunterschiede zwischen unterschiedlichen Bienen-Taxa in der Risikobewertung abdeckt die Notwendigkeit:

- weitere Arten auf ihre Sensitivität gegenüber einer Standardsubstanz (z.B. Dimethoat) zu untersuchen;
- einzelne Wildbienenarten mit einem breiten Spektrum von Pestiziden zu testen.

Darüber hinaus wurden synergistische und antagonistische Effekte von Pestizidkombinationen in den Wildbienenarten *B. terrestris* und *O. bicornis* in Laborstudien nachgewiesen. Daher sollte der potentielle Einfluss von Mischungstoxizitäten in der Effektbewertung sowohl für Wildbienen als auch für andere FVI-Gruppen berücksichtigt werden (Sgolastra et al. 2016; Robinson et al. 2017).

In der Literatur sind mehrere Studien verfügbar, die die toxischen Wirkungen von Pestiziden unterhalb der letalen Dosen (subletale Effekte) auf Wildbienenarten, hauptsächlich Hummeln, untersuchten. Bei umweltrelevanten Dosen konnten in diesen Studien Effekte auf Reproduktion und Nahrungssuche von *Bombus*-Arten festgestellt werden, die in Parametern wie Entwicklung der Oozyten, Eiablage, Nestaufbauaktivität, Koloniegewichtszunahme, Lebenszeit der Arbeiterinnen, Rüsselverlängerungsreflex, Futtersuche oder Blütenbesuch gemessen wurden (z. B. Morandin & Winston 2003; Laycock & Cresswell 2013; Scholer & Krischik 2014; Stanley et al. 2015b; Baron et al. 2017). Darüber hinaus gab es sich widersprechende Ergebnisse zu Kombinationseffekten von Pestiziden und Parasiten (Fauser-Misslin et al. 2014; Piiroinen et al. 2016; Piiroinen & Goulson 2016; Fauser et al. 2017). Neben Hummeln wurden in Laborexperimenten verschiedene andere Wildbienenarten auf subletale Pestizidwirkungen hinsichtlich Parametern wie Larvenentwicklung, Eiproduktion, Überwinterungsvermögen, Geschlechterverhältnis der Nachkommen oder Schlupfrate untersucht (Abbott et al. 2008; Sandrock et al . 2014).

Im Jahr 2013 hat die Europäische Kommission den Einsatz der Neonikotinoid-Verbindungen Imidacloprid, Clothianidin und Thiamethoxam wegen hoher akuter Risiken für Bienen eingeschränkt. Seither wurden mehrere komplexe Feld- und Halbfeldstudien durchgeführt, um das Verständnis von Neonikotinoid-Effekten bei Honig- und Wildbienen in der Agrarlandschaft zu vertiefen. Für diese realistischeren Studien wird angenommen, dass ökologische Unterschiede zwischen Honig- und Wildbienenarten zu unterschiedlichen Reaktionen auf die Belastung durch Pestizide führen (Stoner 2016; Wood & Goulson 2017). In den folgenden Studien wurden die Auswirkungen von Pestiziden auf verschiedene Parameter ermittelt, die für die Erhaltung stabiler Wildbienenpopulationen entscheidend sind. Der größere Teil der Experimente, die Reproduktionseffekte untersuchten, fanden negative Auswirkungen von Pestiziden auf die Entwicklung von Hummelvölkern (z. B. Cutler & Scott-Dupree 2014; Moffat et al. 2015; Moffat et al. 2016; Ellis et al. 2017) oder die Konstruktion von Brutzellen von solitären Bienen (Rundlöf et al. 2015). Zwei große Feldstudien konnten jedoch keinen Einfluss auf die Reproduktion von *B. terrestris* oder *O. bicornis* nachweisen (Peters et al. 2016; Sterk et al. 2016). Neben diesen Effekten auf die Reproduktion beeinflusste eine Behandlung mit Pestiziden auch die Nahrungssuche von Wildbienen in Feld- und Halbfeld-Experimenten. In mehreren Studien konnte ein allgemeines Muster von erhöhter Anzahl und Dauer von Futterflügen, aber verringerter Effizienz der Nahrungssuche gezeigt werden (z. B. Gill et al. 2012; Gill & Raine 2014; Feltham et al. 2014; Stanley et al. 2015a; Stanley & Raine 2016). Nur ein vor Kurzem veröffentlichtes Experiment von Arce et al. (2016) fand lediglich geringfügige Veränderungen in der Nahrungssuche und im Pollensammelverhalten. Effekte auf das Immunsystem von Honigbienen (erhöhte Seuchen- und Parasiten-Anfälligkeit) wurden in mehreren Studien nachgewiesen (z. B. Alburaki et al. 2015; Dively et al. 2015; Pettis et al. 2012; Vidau et al. 2011). Solche Effekte wurden bis heute bei Wildbienen in realen Feldszenarien nicht untersucht.

Andere potentiell relevante Effekte wurden bisher nur selten oder gar nicht untersucht:

- Die Auswirkungen von Formulierungsbeistoffen wurden nur in einigen neueren Studien untersucht, in denen gezeigt wurde, dass die Toxizität dieser Formulierungsbeistoffe und des tatsächlichen Wirkstoffes ähnlich ist und zudem synergistische Wechselwirkungen zwischen Beistoff und Wirkstoff gefunden wurden (Ciarlo et al. 2012; Zhu et al. 2014; Mullin et al. 2015; Fine et al. 2017).
- Indirekte Effekte auf FVI-Populationen durch die Reduzierung der Lebensraumqualität (z.B. durch Reduktion von Nahrungs- und Nistressourcen) wurden bisher kaum untersucht, obwohl die verminderte Diversität und Quantität von Blütenressourcen durch Lebensraumzerstörung und landwirtschaftliche Landnutzungspraktiken als Hauptfaktoren des Rückgangs der Bienen vermutet werden (Müller et al. 2006).
- Der Einfluss von Pestiziden auf die Populationsdynamik auf Landschaftsebene wurde bisher ebenfalls nur selten untersucht. *Source-Sink*-Effekte auf FVI (Meta-)Populationen wurden nur in Landschaftssimulationsstudien von Topping et al. (2014; 2015) betrachtet.
- Bezüglich der von der EFSA (2015) definierten Schutzziele, müssen die Effekte von Pestizidbelastungen auf Ökosystemleistungen wie Bestäubung und Biodiversität noch angemessen untersucht werden. Stanley et al. (2015a) fanden erste Hinweise auf eine verminderte Bestäubung von Apfelbäumen durch *B. terrestris* in einem Halbfeldversuch. Studien zu Auswirkungen auf die biologische Vielfalt sind noch seltener.

Die in der Literatur zur Verfügung stehende Datenbasis bezüglich der Effekte von Pestiziden auf Motten und Schmetterlinge ist im Vergleich zur Datenlage für Bienen deutlich kleiner. Untersuchungen zu direkten toxischen Wirkungen (Mortalität) von Insektiziden auf Lepidopteren haben sich zumeist auf das herbivore Larvenstadium (Raupen) konzentriert (z.B. de Jong et al. 2008; Hoang et al. 2011; Han et al. 2012; Hahn et al. 2015b; Pecenka & Lundgren 2015). In einigen Studien wurden jedoch auch direkte toxische Effekte von Insektiziden auf adulte Lepidoptera berücksichtigt (Salvato 2001; Hoang et al. 2011). Neben letalen Effekten konnten auch verschiedene subletale Reaktionen auf Insektizide festgestellt werden: Gewichtsverlust von Raupen, Veränderungen in der Raupenentwicklung und Verpuppungszeit, Veränderungen in der chemischen Kommunikation und im Paarungsverhalten von adulten Nachtfaltern sowie verminderte Reproduktion (z.B. Clark & Haynes 1992; Abro et al. 1993; Han et al. 2012; Pecenka & Lundgren 2015). Neben Insektiziden können auch Herbizide negative Auswirkungen auf Lepidoptera haben, indem sie das Vorkommen, die Blüte oder die Qualität bestimmter Wirts- oder Nahrungspflanzarten reduzieren (Schmitz et al. 2013; Hahn et al. 2014; Schmitz et al. 2014a, b).

Landwirtschaftliche Pestizidanwendungen können zu letalen und subletalen Effekten auf blütenbesuchende Insekten führen. Dies konnte in Laborstudien vor allem bei Honigbienen, aber auch bei Wildbienen und einigen Lepidoptera-Arten gezeigt werden. Für ein fundiertes Verständnis der Wirkungen von Pestiziden auf blütenbesuchende Organismen besteht jedoch ein Bedarf für weitere Untersuchungen:

- Die akute und chronische Toxizität bei Wildbienen, Lepidopteren (insbesondere Motten) sowie Arten anderer FVI-Gruppen sollten mit einem Fokus auf Wirkungen unterhalb letaler Pestizidkonzentrationen untersucht werden.
- Die Wirkungen anderer Pestizidklassen als Neonikotinoide, Mischtoxizität sowie kombinierte Wirkungen mit anderen Stressoren wie Parasiten und Pathogenen sollten ebenfalls untersucht werden.
- Potentielle Testarten sollten anhand ihrer ökologischen Eigenschaften als Stellvertreter der jeweiligen FVI-Gruppe ausgewählt werden.
- Die Effekte feldrelevanter Pestiziddosen sollten in ökologisch relevanteren Feld- und Halbfreilandversuchen untersucht werden. Hierbei sollte der Fokus auf der Verwendung von FVI-Taxa liegen, die bisher weniger untersucht wurden als Honigbienen oder Hummeln.
- Die toxischen Wirkungen von Fomulierungsbeistoffen sowie die indirekten Effekte von Pestiziden und deren Auswirkungen auf Ökosystemleistungen und Populationsentwicklung von FVIs sollten untersucht werden.
- Feldexperimente sollten so konzipiert werden, dass Rückschlüsse auf die *Source-Sink*-Dynamik von FVI-Populationen in der Agrarlandschaft gezogen werden können. Alternativ könnten *Source-Sink* Effekte in Modellen auf Landschaftsmaßstab simuliert werden. Allerdings sind solche Modelle für FVI noch nicht verfügbar und müssen noch entwickelt und validiert werden (siehe unten).

Empfehlungen für ein FVI Risikobewertungskonzept

Anhand der zusammengetragenen Informationen wurden Empfehlungen für ein Risikobewertungsschema für FVIs abgeleitet. Um dem Vorsorgeprinzip Rechnung zu tragen, sollte als konservativer Ansatz angenommen werden, dass FVIs potentiell auf allen Feldern und in allen Anbaukulturen, auf denen Pestizide angewendet werden, gegenüber Pestiziden exponiert werden. Deshalb sollte für die Autorisation aller Pestizidanwendungen zusätzlich zu einer *off-field* Bewertung auch eine *in-field* Risikobewertung obligatorisch sein. Für die entsprechend ihrer potentiellen Attraktivität für FVIs in Kategorien eingeteilten Anbaukulturen (i.e. attraktive Kulturen, unattraktive Kulturen, vor der Blüte geerntete Kulturen, Wildblumen auf dem Feld) wurden kulturspezifische Szenarien für die FVI-Risikobewertung entwickelt. Hierzu wurden in Abhängigkeit von der jeweiligen Anbaukultur relevante Expositionsszenarien für die verschiedenen Lebensstadien (i.e. Larven, Adulte) der FVI-Gruppen Bienen und Lepidoptera identifiziert. Die entwickelte Matrix soll eine Anleitung bieten für die Auswahl relevanter Risikobewertungsszenarien in Abhängigkeit von der zu bewertenden Pestizidanwendung.

Die meisten FVIs sind hoch mobil und bewegen sich zwischen Nichtziel *off-field*-Flächen und Feldbereichen. Da aufgrund dieser Mobilität negative Effekte auf *off-field* Populationen durch eine Bewertung des potentiellen Risikos auf lokaler Ebene nicht hinreichend protektiv adressiert werden können, sollte die Risikobewertung für FVI nicht nur auf lokalen Maßstab sondern auch auf Landschaftsmaßstab durchgeführt werden.

Für die Risikobewertung auf lokaler Ebene wird ein mehrstufiger Ansatz vorgeschlagen. Auf der ersten Stufe (*Tier-1*) sollte auf Grundlage des aktuellen Wissensstands zu Ökologie und Vulnerabilität von FVIs sowie unter Berücksichtigung der momentan zur Verfügung stehenden Testsysteme ein Basisdatensatz an Toxizitätsdaten gefordert werden:

- 2 Bienenarten (Larven und Adulte), i.e. zwei der vorgeschlagenen Stellvertreterarten.
- 1 Lepidoptera-Spezies (Larven und Adulte), i.e. *Aglais io* oder *Aglais urticae.*

Um das potentielle Auftreten von verzögerten Effekten nach akuter Exposition zu erfassen, sollten neben akuten Effekten auch chronische Effekte in *Tier-1* bewertet werden. Für die Bewertung des Risikos wird entsprechend der entwickelten Matrix für alle als relevant identifizierten Risikobewertungsszenarien die erwartete Exposition in Beziehung zur Toxizität gesetzt werden.

Wenn in den niedrigen Stufen der Risikobewertung ein akzeptables Risiko nicht gezeigt werden kann, besteht die Möglichkeit eines *Refinements* der Bewertung in einer *higher-tier* Risikobewertung auf lokalen Maßstab. Hierfür sollte der Antragsteller anhand von geeigneten *higher-tier* Studien demonstrieren, dass für vulnerable FVI-Spezies kein unakzeptables Risiko besteht. Hierfür stehen folgende *higher-tier* Ansätze zur Verfügung:

- Durchführung von Toxizitätsstudien unter (Semi-)Freilandbedingungen zum *Refinement* substanzspezifischer Aspekte;
- Durchführung von Expositionsstudien zum *Refinement* von Pestizidrückständen in verschiedenen Umweltmatrices (z.B. Nektar, Pollen, Pflanzen, Boden).

Aufgrund der hohen Mobilität vulnerabler FVI-Gruppen, sollten Effekte auf FVI nicht nur auf lokalen sondern auch auf Landschaftsmaßstab betrachtet werden. Um die räumlichen und zeitlichen Variationen in der Dynamik von Pestiziden und die Interaktion mit der räumlichen Dynamik von mobilen FVI zu bewerten, bieten sich Modellierungsansätze auf Landschaftsebene als ein nützliches Werkzeug an. Da die Populationsmodellierung auf Landschaftsmaßstab allerdings sehr komplex ist und zudem mit einem hohen Aufwand verbunden ist, schlägt die Europäische Behörde für Lebensmittelsicherheit (EFSA 2015) die Verwendung von sogenannten *Look-up*-Tabellen als *lower-tier* innerhalb der Risikobewertung auf Landschaftsmaßstab vor. Diese *Look-up*-Tabellen sollen die Ergebnisse von vorab durchgeführten Modellvorhersagen für Standard-Modellierungsszenarien vorhalten. Um die Anforderungen an eine protektive FVI spezifische Risikobewertung zu erfüllen, müssen die von EFSA (2015) vorgeschlagenen *Look-up*-Tabellen für NTA entsprechend an die räumliche Dynamik von mobilen FVI angepasst werden.

Machbarkeitsstudie zur Entwicklung eines *Landscape-Scale Population-Level* Modells für Bestäuber

Landscape-Scale Population-Level Modelle (LSPLM) zur Umweltrisikobewertung sind nicht neu. Allerdings ist deren Anwendung, aufgrund der Schwierigkeiten in der Entwicklung und Pflege der benötigten Modellsysteme, nicht verbreitet. Ein Beispiel für ein bestehendes System zur Modellierung der Auswirkungen von Pestiziden auf Bestäuber auf Landschaftsmaßstab ist das BEEHAVE Modell (Becher et al. 2014). Bei dem BEEHAVE Modell handelt es sich um ein spezifisches Single-Kolonie-Modell für Honigbienen. Zur Modellierung von Populationen solitärer Individuen auf Landschaftsmaßstab ist es daher ungeeignet und als Basis für die Entwicklung von Modellen für andere Bestäuber nicht verwendbar.

Gegenwärtig sind die einzigen Modelle, die alle Populations- und Landschaftsanforderungen erfüllen, unter ALMaSS (Topping et al. 2003) entwickelt. ALMaSS ist ein C++ Modellsystem für die Simulation von terrestrischen Populationen in gemanagten Landschaften. Die Machbarkeit der Entwicklung eines

Modells für Wildbestäuber unter ALMaSS, das für die Risikobewertung von Pestiziden geeignet wäre, wurde im Rahmen dieses Projektes geprüft.

Für die Modellierung geeignete Bestäuberarten sollten folgende grundlegende Voraussetzungen erfüllen:

- um genügend Daten zur Parametrisierung der Modelle zur Verfügung zu haben, sollten die Arten gut erforscht sein;
- benötigte Ressourcen zur Simulation von artspezifischem Verhalten und Ökologie sollten aus räumlich aufgelösten Daten (z.B. kartographierten Daten) vorhersagbar sein;
- die artspezifischen Populationsdynamiken sollten nicht mit anderen Spezies eng verknüpft sein, da dies die zusätzliche Modellierung dieser anderen Spezies erfordern würde. Wobei zu beachten ist, dass dies ein generelles Problem für Bestäuberarten darstellen könnte, deren Populationsdynamik eng verknüpft mit einem Parasiten ist (z.B. Brut- oder Kleptoparasiten bei Bienen).

Sobald grundlegende Informationen zur Verfügung stehen, wie sich Organismen reproduzieren, sterben und sich in der Landschaft bewegen, ist grundsätzlich die Modellierung jeder Spezies möglich. Jedoch müssen für die Akzeptanz eines Modells zur Verwendung für die Risikobewertung von Pestiziden grundlegende Plausibilitätskriterien eingehalten werden. Eines dieser Kriterien ist, dass die zur Parametrisierung und Design des Modells genutzten Daten wissenschaftlich plausible sind. Spezies für die nur wenige oder keine Daten in der wissenschaftlichen Literatur zur Verfügung stehen, werden deshalb als ungeeignet für diesen Ansatz betrachtet. Auf Basis der durchgeführten Literaturrecherche wird die Entwicklung von Modellen für die Bienenarten *Bombus terrestris und Osmia bicornis* sowie die Lepidoptera-Spezies *Maniola jurtina* als machbar betrachtet.

Risikominderungsmaßnahmen

Zum Schutz von FVI vor pestizidbedingten Effekten können Risikominderungsmaßnahmen implementiert werden. Hierzu wurden insgesamt 13 anwendungs- oder landschaftsbezogene Maßnahmen vorgeschlagen und hinsichtlich folgender Aspekte evaluiert:

- Effektivität der Maßnahmen die Pestizidexposition von FVIs zu reduzieren;
- Effektivität der Maßnahmen Artenreichtum und Abundanzen von FVIs in der Agrarlandschaft zu verbessern;
- Durchführbarkeit und Akzeptanz der Maßnahmen durch Landwirte.

Maßnahmen wie z.B. *in-field* Pufferstreifen, *no-spray* Zonen oder Brachen können dazu beitragen, Pestizideinträge in *off-field* Habitate zu reduzieren und infolgedessen die Pestizidexposition von FVI auf diesen Nichtzielflächen reduzieren. Darüber hinaus können anwendungsbezogene Maßnahmen die Exposition von FVI auf *in-crop* Flächen reduzieren (z.B. Verringerung der Aufwandmenge, Anwendung in Zeiten in denen tagaktive FVI nicht aktiv sind). Landschaftsbezogene Maßnahmen (z.B. *in-field* Pufferstreifen, Vergrößerung kleiner Ackerrandstreifen, Management von *off-field* Habitaten) können durch Änderungen in der Landschaftsgestaltung zu einer Verbesserung der Verfügbarkeit und/oder Qualität von FVI-Habitaten führen. Damit haben landschaftsbezogene Maßnahmen das Potential die Abundanz und das Artenreichtum von FVIs in der Agrarlandschaft zu fördern und pestizidbedingte Effekte auf FVI Gemeinschaften möglicherweise zu kompensieren.

Die hinsichtlich ihrer Effektivität FVI Gemeinschaften zu fördern, am besten untersuchten Risikominderungsmaßnahmen sind *in-field* Pufferstreifen mit Wildblumen und Brachen. Mit Wildblumen (oder mit Nektar und Pollen Mischungen) angesäte *in-field* Pufferstreifen steigern am effektivsten die Abundanzen und Artenvielfalt von Bienen, Schmetterlingen und Schwebfliegen. Wohingegen andere Streifenelemente (z.B. mit natürlicher Vegetation oder mit Gras bewachsene Pufferstreifen) weniger effektiv für Bienen und Schmetterling Populationen zu sein scheinen. Ganz grundsätzlich stellt die Anzahl der zum Fouragieren zur Verfügung stehenden Blüten einen wesentlichen Faktor für die Effektivität von Blühstreifen dar. Zur Anlage von Blühstreifen sollten regionale Samenmischungen indigener Pflanzen benutzt werden. Darüber hinaus sollten die Samenmischungen Schlüsselpflanzenarten enthalten, um das Vorkommen von auf bestimmte Pflanzen spezialisierter FVI Taxa zu fördern. Hinsichtlich der Größe von Streifenelementen kann davon ausgegangen werden, dass mit zunehmender Streifenbreite und damit wachsender Anzahl an Futter- und Nistressourcen die FVI Abundanzen ansteigen.

Für Naturschutzbrachen wurden positive Effekte auf Hummeln, Wildbienen, Wespen, Schmetterlinge und Motten in der wissenschaftlichen Literatur nachgewiesen. Ähnlich wie für *in-field* Pufferstreifen werden sowohl die Diversität der Blüten als auch der Bedeckungsgrad mit Blüten als wichtige Faktoren für die Abundanz und Diversität von FVI Gemeinschaften beschrieben. Die Auswirkungen von durch natürliche Regeneration entstandenen Brachflächen wurden bisher nicht umfassend untersucht. Jedoch gibt es Hinweise darauf, dass aufgrund des hohen Grads an Diversität der Pflanzenlebensgemeinschaft natürlich regenerierte Brachen ein effektives Werkzeug zum Schutz von FVIs sein können. Hinsichtlich der Lebensdauer von Brachen lässt sich zusammenfassen, dass mehrjährige Brachen ein breiteres Spektrum an Ressourcen zur Verfügung stellen und so auch schon früh in der Saison Nahrungshabitate bieten können. Speziell FVI Gruppen, die schon früh in der Saison aktiv sind, können von diesem Angebot an Blüten im Frühling profitieren.

Neben *in-field* Blühstreifen und Brachen konnte die Effektivität FVI-Lebensgemeinschaften in der Agrarlandschaft zu fördern, für weitere Risikominderungsmaßnahmen demonstriert werden (z.B. Hecken, Aussaat von Samenmischungen, *no-spray* Zonen). Jedoch besteht vor einer abschließenden Bewertung für viele der evaluierten Maßnahmen weiterer Forschungsbedarf. Insbesondere sind für anwendungsbezogene Maßnahmen (z.B. Verringerung der Aufwandmenge, spraydriftreduzierende Techniken) keine Studien in der Literatur verfügbar, die potentielle Effekte auf FVIs untersuchen. Im Fokus der verfügbaren Studien zu diesen Maßnahmen ist deren Potential entweder Pestizideinträge in *off-field* Habitate oder auf der Anbaufläche zu reduzieren. Allerdings kann, aufgrund ihrer nachgewiesenen Effektivität Pestizidexpositionen zu verringern, angenommen werden, dass diese Maßnahmen auch zu einer Reduktion von pestizidbedingten Effekten auf FVI führen. Aber nicht nur für anwendungsbezogene Maßnahmen, sondern auch für einige landschaftsbezogene Maßnahmen (i.e. Schaffung von Nistplätzen, Belassen von Totholz in Obstplantagen, Pflege von Hecken) ist die in der Literatur verfügbare Datenbasis für eine Bewertung hinsichtlich FVIs unzureichend. Jedoch kann basierend auf ökologischen Gesichtspunkten die Effektivität dieser Maßnahmen zur Förderung von FVI Gemeinschaften angenommen werden.

Aufgrund der hohen Mobilität der meisten FVI-Taxa ist der langfristige Fortbestand von FVI Populationen in der Agrarlandschaft abhängig von einem Gleichgewicht zwischen Quellpopulationen (z.B. in semi-natürlichen Habitaten) und Populationssenken (z.B. auf landwirtschaftlichen Nutzflächen oder in durch Pestizide beeinflussten *off-field* Habitaten). Um den Fortbestand von FVI Populationen zu fördern, sollte die Implementierung von Risikomanagementmaßnahmen deshalb darauf abzielen, Quellpopulationen zu stärken und/oder Populationssenken zu verringern.

Möglichkeiten zur Stärkung von Quellpopulationen:

- Erhalt von großen off-field Habitaten in der Agrarlandschaft (z.B. Graslandschaften, Wäldern, Naturschutzgebieten);
- Management bestehender off-field Habitate mit dem Ziel der Vergrößerung von Pollen und Nektar Ressourcen;
- Schaffung einer heterogenen Landschaft durch die Implementierung linear strukturierter Streifenelemente (z.B. Blühstreifen) und flächig strukturierter Elemente (z.B. Brachen).

Möglichkeiten zur Verringerung von Populationssenken:

- Reduktion von Pestizidanwendungen hinsichtlich z.B. Anwendungsrate, -intervall und -frequenz;

- Verwendung spraydriftreduzierender Techniken;
- Timing der Pestizidanwendung;
- Management der Vegetation in Populationssenken hinsichtlich einer Verbesserung der Qualität von bestehenden Habitaten sowie der Schaffung neuer Habitate mit dem Ziel Ressourcen für Wachstum, Erhalt und Reproduktion von FVI Populationen zur Verfügung zu stellen.

Dabei ist allerdings zu beachten, dass manche FVI-Gruppen ein inverses Muster von der normalerweise zu erwarteten Quellen-Senken-Beziehung aufweisen. Deshalb sind für eine angemessene Vorhersage der Konsequenzen der Implementierung von Risikominderungsmaßnahmen auf die Populationsdynamiken der zu schützenden Spezies ausreichend Informationen bezüglich ihres Lebenszyklus sowie ihren spezifischen Habitatsanforderungen wichtig.

Ökologische Vorrangflächen

Die gemeinsame Agrarpolitik der EU (*common agricultural policy*, CAP) wurde im Jahr 2013 unter anderem mit dem Ziel der Stärkung einer nachhaltigen, ökologischen Landwirtschaft reformiert. Hierzu wurde eine sogenannte *Greening*-Komponente bei den Direktzahlungen eingeführt. Das *Greening* umfasst unter anderem die Ausweisung von ökologischen Vorrangflächen (*ecological focus areas*, EFAs), die dem Erhalt der Biodiversität und der natürlichen Ressourcen dienen sollen. Danach sind Landwirte mit mehr als 15 ha landwirtschaftlicher Nutzfläche verpflichtet, mindestens 5% ihrer Ackerflächen als ökologische Vorrangflächen vorzuhalten und entsprechend zu bewirtschaften, um Direktzahlungen aus der ersten Säule der gemeinsamen Agrarpolitik der EU zu erhalten. Für dieses F&E-Projekt wurden die verschieden EFA-Typen anhand der zur Verfügung stehenden wissenschaftlichen Literatur evaluiert, zum einem hinsichtlich ihres Potentials Pestizideinträge in benachbarte *off-field* Habitate zu reduzieren und zum anderen hinsichtlich ihrer Effektivität die Abundanzen und das Artenreichtum von FVI-Gemeinschaften in der Agrarlandschaft zu verbessern. Für fünf EFA-Typen (i.e. Brachen, Hecken, Feldsäume, Pufferstreifen und Streifenelemente entlang von Waldrändern) konnte die Effektivität FVIs zu fördern als wissenschaftlich belegt klassifiziert werden. Für alle anderen EFA-Typen ist die in der Literatur verfügbare Datenbasis für eine Bewertung hinsichtlich FVIs unzureichend. Jedoch können basierend auf ökologischen Gesichtspunkten positive Effekte auf FVI-Gemeinschaften erwartet werden.

Im Rahmen des vorliegenden Berichts wurde die Frage diskutiert, welche Mindestanforderungen für EFAs gelten, damit diese Flächen einen positiven Beitrag zum Schutz von FVI-Lebensgemeinschaften leisten können. Wichtige Aspekte hierbei sind die Größe der Fläche, die Anzahl der EFAs in der Agrarlandschaft und ihre Vernetzung sowie die Lebensdauer des Biotops. Bezüglich der Größe von EFA-Flächen kann angenommen werden, dass mit zunehmender Größe mögliche negative Randeffekte abnehmen (z.B. Randeffekte durch Pestizideinträge via Drift). Darüber hinaus können größere Flächen mehr Futter- und Nistressourcen bereitstellen. Jedoch konnte gezeigt werden, dass FVI auch schon von der relativ schmalen Breite von Streifenelementen profitieren. Anhand der verfügbaren Daten konnte für an Feldkulturen angrenzende Pufferstreifen (natürlich regeneriert oder mit Wildblumen/Gräser bewachsen) eine Mindestbreite von 6 m abgeleitet werden, um eine Vielzahl an FVI-Spezies zu unterstützen. Für Brachen konnte eine Mindestfläche von 0.6 ± 0.4 ha abgeleitet werden. Neben der Größe scheint das Alter der Flächen ein wichtiger Faktor zu sein: So scheinen FVI-Gemeinschaften von einer Mischung aus ein- und mehrjährigen Pufferstreifen bzw. Brachen in der Agrarlandschaft zu profitieren.

Der Anteil von EFAs an der gesamten Landwirtschaftsfläche, der notwendig wäre um pestizidbedingte negative Effekte auf FVI Populationen ausreichend zu kompensieren, ist auf Basis der zurzeit zur Verfügung stehenden Daten nur schwierig zu bestimmen. Aktuelle Studien gehen von einen prozentualen Anteil von 3-7% natürlicher Habitate (Cormont et al. 2016) oder 7.5% nicht für landwirtschaftliche Produktion genutzter Fläche (Holland et al. 2015) aus. In einer weiteren Studie konnte gezeigt werden, dass die Aufwertung von 10% der Flächen z.B. als Blühstreifen oder Brachen in einem 50 ha großen Untersuchungsgebiet eindeutig die Diversität und Abundanz von FVIs förderte (Oppermann et al. 2016; Maus et al. 2017). Aufgrund der sehr kleinen Datenbasis ist aber ein Mindestanteil nicht eindeutig ableitbar. Allerdings scheint neben der reinen Quantität die Qualität der EFAs (i.e. Qualität der Futter- und Nistressourcen) eine wichtige Rolle zu spielen.

Das Hauptziel von EFAs ist der dauerhafte Schutz und die Förderung der Biodiversität in der Agrarlandschaft, dass schließt unter anderem auch die Diversität von FVIs mit ein. Da die Anwendung von Pestiziden in der Agrarlandschaft im Verdacht steht, eine Ursache für den Rückgang der Artenvielfalt zu sein (Marshall & Moonen 2002; Brittain et al. 2010; Balmer et al. 2013; BMEL 2013), sollten - um das definierte Ziel (Erhaltung der Biodiversität) erfüllen zu können - Pestizide in EFAs nicht angewendet werden. Darüber hinaus sollten die EFAs vor Pestizideintrag (z.B. via Spraydrift) geschützt werden. Für einen effektiven Schutz der EFAs bieten sich verschiedene Optionen an:

- Errichtung von *no-spray* Zonen angrenzend an die EFAs;
- Implementierung unkultivierte Pufferstreifen angrenzend an die EFAs (z.B. wie in der Schweiz). Zur Erhöhung der Akzeptanz bei Landwirten sollte die Anlage und Pflege solcher Pufferstreifen finanziell gefördert werden, um den Verlust von Anbaufläche und infolgedessen den Ertragsverlust zu kompensieren;
- Verwendung von driftreduzierenden Düsen oder Enddüsen auf an EFAs angrenzenden Flächen.

Fördermöglichkeiten für Risikomanagementmaßnahmen

Im Rahmen der gemeinsamen Agrarpolitik der Europäischen Union stehen zur Unterstützung von Landwirten via Direktzahlungen (1. Säule) und zur Förderung der nachhaltigen und umweltschonenden Bewirtschaftung und Entwicklung ländlicher Regionen (2. Säule) EU-Mittel zur Verfügung. Als Teil der 1. Säule spielt die *Greening*-Prämie eine wichtige Rolle zur Förderung von Risikomanagementmaßnahmen. Das *Greening* umfasst die Maßnahmen Anbaudiversifizierung, Erhalt von Dauergrünlandflächen (Wiesen und Weiden) und die Bereitstellung ökologischer Vorrangflächen. Einige der zum Schutz von FVI s in der Agrarlandschaft in diesem Bericht vorgeschlagenen Risikominderungsmaßnahmen (z.B. Brachen oder Pufferstreifen) können im Rahmen der *Greening*-Prämie als ökologische Vorrangflächen gefördert werden.

Die 2. Säule der Förderung umfasst spezifische Programme zur nachhaltigen und umweltschonenden Landwirtschaft und ländlicher Entwicklung. Zentrales Förderinstrument bei der Umsetzung der EU-Prioritäten für die Entwicklung ländlicher Regionen ist der Europäische Landwirtschaftsfonds für die Entwicklung des ländlichen Raums (EAFRD). Jedem EU Mitgliedsstaat werden für die Bereitstellung von EAFRD Programen Mittel aus diesem Fonds zugeteilt. Für den Förderzeitraum von 2014-2020 werden in Deutschland 13 Programme für ländliche Entwicklung (RDP) auf Ebene der Bundesländer durchgeführt. Im Zentrum dieser Programme stehen freiwilligen Agrarumwelt- und Klimaschutzmaßnahmen der Landwirtschaft, sowie Maßnahmen zur Verbesserung des Tierschutzes und Förderung des Ökologischen Landbaus. Die Agrarumweltmaßnahmen der RDPs von Rheinland-Pfalz (EULLE 2015), Sachsen-Anhalt (EPLR 2015) und das gemeinsame Programm von Niedersachsen und Bremen (PFEIL 2015) wurden exemplarisch auf mögliche Fördermöglichkeiten für die Implementierung FVI-spezifischer Risikominderungsmaßnahmen überprüft. Die Auswertung zeigte, dass Maßnahmen wie *in-field* Pufferstreifen und das Management von *off-field* Habitaten (z.B. Mahd-Rhythmus von Grünlandflächen) in allen drei Bundesländern gefördert werden. Während die Förderung anderer Maßnahmen bundeslandspezifisch ist (z.B. Anlage und Pflege von hochwachsender Vegetation wie z.B. Hecken oder das Belassen von Totholz in Obstplantagen).

Abschließend wurden bestehende Wissenslücken identifiziert und es wurden Vorschläge für weitere Forschung skizziert, die einen Beitrag zur Vertiefung unseres Verständnis der Effekte von Pestiziden auf FVIs und zur Verbesserung existierender regulatorischer Risikobewertungsverfahren leisten soll.

1 Deficit analysis

Philipp Uhl, Carsten Brühl

1.1 Introduction

Pesticides are applied in crops to reduce pest pressure and thus increase yield. However, not only pests but also non-target species are exposed to these chemicals. Through multiple pathways, such as root uptake and subsequent translocation in the plant or direct overspray, flowers of crops and weeds may be contaminated with pesticides that might be consequently expose flower-visiting insects (FVIs).

Many FVIs are vital pollinators of crops and wild plants in the European agricultural landscape. Non-bee FVIs include flies, beetles, moths, butterflies, wasps and ants, many of which are also important pollinators (Rader et al. 2015). Plant pollination is a central ecosystem service since 35% of global agricultural production is related to crops that increase yield when pollinated by animals (Klein et al. 2007). The global economic value of animal pollination was estimated to be €153 billion (Gallai et al. 2009). However, biotic pollination is not just economically important but also essential for the preservation of native flora since 85% of all flowering plants are pollinated by animals (Ollerton et al. 2011). Flowering agricultural crops and wild plants are mostly pollinated by insects (Klein et al. 2007; Ollerton et al. 2011). However, not all crops are dependent on insect pollination because they are self-fertile. Some crops (e.g. cereals) do not require insect pollination since they are wind-pollinated or non-fruit parts are consumed (e.g. potatoes, carrots) (Ghazoul 2005).

There is growing evidence that global FVI numbers are decreasing because of environmental change, related to factors such as climate change, habitat loss, environmental pollution and pesticide use (Goulson et al. 2015; Hallmann et al. 2017). This development has been most prominent in bee species. In Germany 53% of bee species are red listed (Westrich et al. 2011), in some European countries even up to 65% (Patiny et al. 2009). The European red list of bees documents 9% of bee species as threatened and additional 5% as near threatened. For 57% of all European bee species there is not enough data to determine their threat status. The actual proportion of threatened bee species was estimated as up to 61% (Nieto et al. 2014). Possible causes are habitat loss and fragmentation, pesticides and environmental pollution, decreasing resource diversity, invasive species, pathogens and climate change (Goulson et al. 2015). Since the last century the USA and Europe are experiencing substantial losses of domestic honey bee (*Apis mellifera*) hives and simultaneous decline in wild bee diversity (Natural Research Council 2006; vanEngelsdorp et al. 2008; Potts et al. 2010, 2015; Goulson et al. 2015). Furthermore, there are population declines of butterfly, moth and syrphid fly species recognised in the EU (EASAC 2015; Gilburn et al. 2015; Potts et al. 2015). A recent study showed a decline in biomass of insects in nature reserves in the German agricultural landscape over 27 years by more than 75%, including many FVIs such as butterflies, bees, flies and beetles (Hallmann et al. 2017). This development is driven by the abovementioned factors and agricultural pesticide use (EASAC 2015; Gilburn et al. 2015; Potts et al. 2015; Forister et al. 2016). The honey bee is the main pollinator species employed by humans (Klein et al. 2007; Winfree et al. 2007). However, it has become clear that regarding the ecosystem service pollination focusing on only one species is not advantageous. Wild insects often provide more effective pollination services. In a meta-analysis of 41 crop systems worldwide wild FVIs were found to be qualitatively better pollinators than the honey bee, i.e. less pollen deposition on the stigma was necessary to initiate fruit set. Furthermore, fruit set increased with wild FVI flower visit in all analysed crops systems but only in 14% with honey bee visits (Garibaldi et al. 2013). Hence, honey bees can only complement pollination by native insects but cannot replace it. Furthermore, pollinator diversity increases fruit set in many arable and orchard crops worldwide (Garibaldi et al. 2013; Mallinger et al. 2015). However, not all FVI species are quantitatively relevant pollinators or have not yet been recognised as such (Orford et al. 2015; Grass et al. 2016; Hahn & Brühl 2016).

1.2 A definition of flower-visiting insect species

FVI species are threatened and pesticides are a recognised contributing factor for this development (EASAC 2015; Gilburn et al. 2015; Goulson et al. 2015; Potts et al. 2015). This is most noticeable in bee species, especially honey bees since they are important pollinators and there is an economic interest in preserving viable populations (Klein et al. 2007; Gallai et al. 2009). However, there are many other flower-visiting taxa that might be exposed to pesticides in the agricultural landscape and consequently be affected (Godfray et al. 2014, 2015; Gilburn et al. 2015). Hence, the terms “pollinator” and “flower-visiting insect” are to be distinguished and the whole community of FVI species should be addressed when investigating the impact of agricultural pesticide use. However, there are novel approaches that incorporate ecological traits to model and predict the pollinator potential of FVI species that might be applied in future studies of this functional group (Coetzer et al. 2016; Stavert et al. 2016). In the context of this report the term FVI is referring to insect taxa that forage on flower resources such as nectar and pollen in at least one life stage. The relevant groups consist of species that have a considerable chance to be exposed to pesticides.

Bees (Apiformes)

All bee species (1965 in Europe) are obligate florivores in larval and adult life stages. This distinguishes them from all other FVI taxonomic groups where only a subset of species are flower visitors and only adults are florivores. Adult bees feed on pollen and nectar and larvae are feed mostly pollen but also nectar (Michener 2007; Winfree et al. 2011; Nieto et al. 2014).

Flies (Diptera)

This diverse taxonomic group of globally over 150000 species is recognised as the second most important flower visitors (Larson et al. 2001; Winfree et al. 2011). FVI species are found in three families: Syrphidae (hover flies/syrphid flies), Bombyliidae (bee flies) and Tachinidae (tachinid flies). Hover flies are considered the key group (c. 800 species in Europe) where nearly all species' adults consume nectar and sometimes also pollen (Larson et al. 2001; Winfree et al. 2011). However, this statement might need re-evaluation since recent findings by Grass et al. (2016) and Orford et al. (2015) show that a substantial part of FVI species in wildflower plantings and farmland are dipterans other than hover flies.

Moths & Butterflies (Lepidoptera)

Butterflies account only for a small part of lepidopterans as opposed to the general opinion. Moth are the predominant group of this taxonomic order as make up 95% of ca. 10000 European lepidopteran species (Swaay et al. 2010). The majority of lepidopteran species are nectarivorous, only very few consume pollen. Flower-visiting species are mostly part of the moth family Noctuidae (owlet moths) and the butterfly superfamily Papilionoidea (common butterflies). Furthermore, there are some European species of Sphingidae (hawk moths) and Hesperiidae (skippers) (Winfree et al. 2011). Not much is known about the interactions of moths with flowering plants. They might actually be relevant as flower visitors, especially for non-crop plants (Hahn & Brühl 2016).

Wasps/Beetles (Vespinae/Coleoptera)

There are flower-visiting wasp species in the families Vespidae, Scoliidae, Pompilidae and Agaonidae. Furthermore, there are also beetle species from multiple families that are recognised as pollen feeders. However, data on the prominence of these taxa as FVIs is scarce (Winfree et al. 2011). Wardhaugh (2015) estimated that approximately 30 % of the global arthropod species are regular flower visitors. Therefore, several taxa might be quantitatively relevant FVIs but cannot be recognised as such due to an insufficient database.

The complexity of plants and their insect visitors was shown in a recent network analysis in German meadows including more than 25000 interactions between 166 plant species and 741 pollinator species (Weiner et al. 2014). The network incorporated 115 bee species, including 25 pollen specialists (oligolectic bees), 48 other hymenopterans, 50 butterflies, 104 beetles, 103 syrphids, and 321 other dipteran species.

1.3 The European risk assessment scheme for FVI species

The aforementioned contribution of pesticides to the decline in many groups of FVI species suggests that FVIs are not sufficiently protected from pesticide effects under the current European risk assessment and risk management schemes. This might be the result of uncertainties in existing risk assessment practice which seems to be the case for substance classes such as neonicotinoids where negative direct effects were observed especially for bees but also for flies and lepidopterans (Godfray et al. 2014, 2015; Gilburn et al. 2015). These insecticides have been shown repeatedly to cause adverse effects in field studies on certain FVI species when applied at approved field rates. Furthermore, they have been linked to population declines of FVI species in scientific reviews (Godfray et al. 2014, 2015; Pisa et al. 2015; Rundlöf et al. 2015). After some incidences with honey bee mortality, the three most prominent neonicotinoids (imidacloprid, thiamethoxam, clothianidin) were restricted in use and sale by the EU because of their potentially high risks towards bees (European Commission 2013). This emphasises deficits in the European pesticide registration process concerning bees and possibly all FVI species (Klatt et al. 2016). The assessment of the risk from the use of pesticides to FVIs is currently implemented by applying two separate proxies in EU legislation: Bees and non-target arthropods NTAs other than bees.

Under Regulation (EC) No 1107/2009 of the European Parliament and of the Council a pesticide can only be registered and placed on the European market if it "will result in a negligible exposure of honeybees" or has "no unacceptable acute or chronic effects on colony survival and development, taking into account effects on honeybee larvae and honeybee behaviour". Furthermore, the "possibility of exposure of beneficial arthropods other than honeybees" is to be investigated and "lethal and sublethal effects on these organisms" are to be assessed (European Commission 2009). The risk assessment schemes and associated data requirements for both groups are specified in Regulation (EU) No 283/2013 (active substances), No 284/2013 (formulations), as well as the Commission communications C 95/1 (active substances) and C 95/21 (formulations) by referring to different guidance documents (European Commission 2013c, a, d, b).

Bee risk assessment

The currently valid guidance document on Terrestrial Ecotoxicology under Council Directive 91/414/EEC (SANCO 2002) refers to a protocol of the European and Mediterranean Plant Protection Organization for bee risk assessment schemes (OEEP/EPPO 2010a, b). Within this scheme several tests are recommend to evaluate acute and chronic direct effects of pesticides in lab and field scenarios, however the entire spectrum of FVI species is not well-represented since the honey bee (*Apis mellifera*) is used as the only surrogate for all bee species (OEEP/EPPO 2010b). It has been criticised that wild bee species such as bumble bees and solitary bees have different ecological properties, e.g. sociality, life cycle, behavior, and therefore their sensitivity towards pesticides might differ substantially from the honey bee (Arena & Sgolastra 2014; Rundlöf et al. 2015; Stoner 2016). In bee tier I testing the effects of oral and contact exposure of honey bees to pesticides are evaluated in the laboratory. Exposure assessment for bees considers direct overspray, consumption of contaminated plant material (nectar/pollen) and (guttation) water, as well as drift of pesticide dust (European Commission 2013c, d). However, exposure by dust or guttation water is not implemented in the current risk assessment scheme (Table 1). Contact exposure is estimated by using the application rate and oral exposure by using data from residue and plant metabolism studies (OEEP/EPPO 2010a, b). Additionally, exposure of non-target areas and non-target plants is not accounted for. In higher tier testing several more complex test systems with more realistic exposure scenarios can be used to refine risk assessment: Brood feeding tests, semi-field studies using tunnel tents or field tests. The design of semi-field and field studies has been criticised for allowing for too much variance in the collected data and providing not enough statistical power due to e.g. small tunnels, high genetic variance between colonies, low sample size and number of colonies per field/tunnel or insufficient space between field sites to avoid foraging on between treatments (EFSA PPR Panel 2012; EFSA 2013). Higher tier exposure assessment does include pesticide residue field studies of relevant matrices such as dead bees, nectar, pollen, wax or honey (OEEP/EPPO 2010a, b).

NTA risk assessment (except bees)

The Guidance Document on Terrestrial Ecotoxicology under Council Directive 91/414/EEC (SANCO 2002) applies also to NTAs. Here the risk assessment refers to results of the Escort 2 workshop (Candolfi et al. 2001). The cereal aphid parasitoid *Aphidius rhopalosiphi* and the predatory mite *Typhlodromus pyri* are used as surrogates in tier I testing. Acute and chronic contact laboratory tests are performed. Contact exposure is simulated by putting test individuals on oversprayed glass plates, plant material or soil. Both are no FVI species but predatory or parasitic arthropods that are used in biological control schemes of pest mites and aphids. Hence, all non-bee FVI species are not accounted for by these surrogate organisms (Candolfi et al. 2001). Higher tier testing includes extended lab and age residue studies, semi-field and field experiments. For these studies four additional test species are proposed but do also not include FVIs but beneficial predatory arthropods historically derived from integrated pest management. NTA exposure assessment is performed separately for in-field overspray and off-field spray drift. Only contact toxicity is tested which means that the relevant pathway of exposure consuming contaminated nectar, pollen or guttation water is not covered (Table 1). Furthermore, exposure assessment of dust drift after sowing of pesticide-treated seeds is also not performed. For higher tier testing more complex tests are performed from extended lab studies to field experiments. Semi-field and field studies allow for an assessment of both contact and oral exposure that usually contain natural arthropod communities monitored with specific trapping methods.

1.4 Development of new guidance documents by EFSA

The current risk assessment schemes for bees and NTAs are showing clear deficits regarding the specific exposure and effect profiles of FVIs (Table 1).

In 2012 EFSA published a scientific opinion on the science behind the development of a risk assessment of Plant Protection Products on bees (*Apis mellifera, Bombus* spp. and solitary bees) in which the current test guidelines were evaluated and certain improvements were suggested (EFSA PPR Panel 2012). This lead to the preparation of the “Guidance on the risk assessment of plant protection products on bees (*Apis mellifera, Bombus* spp. and solitary bees)” (EFSA 2013). This new guidance document defines Specific Protection Goals (SPGs) based on an ecosystem service approach. Bees are to be protected as:

- Pollinators of in-field crops and off-field non-target wild plant species
- Supplier of food provision services (honey)
- Genetic resource with educational and aesthetic value

The rather straightforward first SPG is not protective for all FVIs because not all of them are quantitatively relevant pollinators (see above, flower visits are not necessarily resulting in successful pollination). Moreover, FVIs generally do not produce honey which is a unique feature of honey bees. The concept of educational or aesthetic value is rather vague and can be applied to every living creature. It is therefore unclear how to evaluate such values in FVI communities. Protection of genetic resources seems rather subjective and is simply a substitute for the concept of biodiversity which scientists and administrators still struggle to put in an ecosystem service context (EFSA PPR Panel 2012; Mace et al. 2012).

The exposure assessment has been adjusted to incorporate more likely uptake pathways of pesticides for bees such as dust from seed treatment and guttation water from seedlings emerging from treated seeds (Table 1). Off-field exposure is also incorporated via a deposition factor for spray, granular and seed treatment application. There have also been changes in effect assessment due to recent scientific research: Arena & Sgolastra (2014) performed a meta-analysis of acute toxicity studies on wild bees' reaction to different insecticides of common pesticide classes compared to the honey bee. They could show that in most cases wild bees are less sensitive. However, the dataset needs to be treated with caution for European FVIs since endpoints for 9 out of 19 species came from tropical, mostly social bees and 5% of wild bee species were more sensitive than the honey bee. This was consistent for all tested classes of insecticides except for neonicotinoids. Towards these substances wild bees showed equal to higher sensitivity than the honey bee. Overall, there still remains reasonable doubt that the honey bee is an adequate surrogate organism to cover differences in species sensitivity at least under field-realistic conditions due to severe differences in ecological properties compared to most wild bee species (e.g. sociality, life cycle, behaviour) (Arena & Sgolastra 2014; Cutler et al. 2014; Rundlöf et al. 2015). To better reflect differences in species' sensitivity in bee risk assessment EFSA (2013) proposed two additional test species: The buff-tailed bumble bee *Bombus terrestris* and the solitary red mason bee *Osmia bicornis* (syn. *Osmia rufa*). These wild bee species should improve bee risk assessment because of their different sociality and life history throughout the year which may lead to different responses to pesticides than in the honey bee. However, it is unclear if these new test species are suitable in this respect. Both species are for example less sensitive to the insecticide dimethoate, which is often used as positive control treatment or toxic reference, than the honey bee in acute laboratory (tier I) tests (Uhl et al. 2016). Nonetheless, it may still be reasonable to use these species for higher tier (e.g. semi-field studies) testing where ecological differences such as sociality influence toxicity to a greater extent (Cutler et al. 2014; Rundlöf et al. 2015). Arena & Sgolastra (2014) further proposed that honey bee endpoints may be used for bumble bee and solitary bee effect assessment using a bridging factor of 10. If bumble bee or solitary bee testing is actually performed this bridging factor will be dropped and an assessment factor of 5 is suggested to account for uncertainties in interspecific sensitivity. However, reducing such a bridging factor after testing just one representative species might not prove protective since uncertainty may only be substantially reduced after testing multiple species. In lower tier effect assessment EFSA proposed to add a chronic oral and a chronic larval toxicity test. In higher tier effect assessment EFSA suggested to modify the study design to decrease data variance and enhance statistical power. For honey bee field and semi-field studies this includes larger tunnel/field size, higher number of replicates and colonies per site, greater distance between sites, the use of sister queens in colonies and prolonged study duration. Similar designs were proposed for *O. bicornis* and *B. terrestris*. However, so far there are no established semi-field or field test protocols for both species. Furthermore, the ambitious study design improvements by EFSA (2013) might be difficult to implement. Therefore, ringtests are currently conducted to the evaluate the feasibility of these improved study designs for the three test species (e.g. International Commission for Plant-Pollinator Relationships solitary bee higher tier ringtest (Knäbe et al. 2017)). For residue studies in higher tier exposure assessment EFSA suggested to include pesticide doses in plants or bees foraging on the treated crop as well as bees returning to the hive (EFSA 2013). The bee guidance document has yet to be ratified by the EU member states due to knowledge gaps and uncertainties concerning wild bee species and their sensitivity. The European Commission drafted a "roadmap" for the implementation which specifies areas that need further studying. The process of revision and implementation should be finished by 2019 (European Commission 2014).

EFSA did also recently publish a "Scientific Opinion addressing the state of the science on risk assessment of plant protection products for non-target arthropods" (EFSA 2015). This opinion is the precursor of an upcoming new non-target arthropod (NTA) guidance document. As in the bee guidance document ecosystem service-based SPGs were defined. Pollination is also explicitly mentioned here, overlapping with the bee guidance document. However, according to the NTA opinion population effects should be negligible in off-field areas whereas in-field small effects on NTA populations over months are accepted. In-field the functional group is the reference i.e. to guarantee sufficient pollination of crops. Since most FVIs are highly mobile species and pollination service is often driven by a few dominant species (Kleijn et al. 2015; Senapathi et al. 2015) it is questionable if both goals are generally compatible. Therefore, EFSA called for a landscape-scale risk assessment for mobile species that are likely to switch between in-field and off-field areas. This should ensure that in-field effects do not lead to unacceptable reductions in off-field populations (EFSA 2015). The ecosystem services biodiversity/genetic resources and cultural services (aesthetic value) target the protection of populations but are difficult to define as mentioned above. For example a moth might be not as appealing as a butterfly but still be exposed to pesticides while visiting wild flowers in a field margin (Hahn & Brühl 2016). Exposure assessment has been improved since pesticide uptake through food (nectar, pollen, plant material) and dust was included (Table 1). Furthermore, one FVI (lepidopteran larvae) has been proposed as an additional test species for effect assessment to represent a herbivore species. This is reasonable since lepidopterans have been shown to react to field margin-realistic pesticide doses with lethal and sublethal effects and are therefore not adequately protected up until now (Hahn et al. 2014, 2015b). However, due to the multitude of different life strategies and ecological niches of non-bee FVIs it remains unclear if one test species will be sufficient to substantially improve risk assessment for this group, especially in higher tier assessment. Furthermore, the NTA scientific opinion is lacking in concrete protocols for effect and exposure assessment. The new NTA guidance document should therefore ideally follow up with more tangible recommendations.

Additionally, the current risk assessment scheme does not address indirect effects of pesticides. Since FVIs depend on plant resources, a reduction in plants through herbicide use might have an indirect effect. In a study of a natural plant community herbicide application rates corresponding to realistic input in field margins next to cereal fields resulted in shifts in plant community composition (Schmitz et al. 2014a). Additionally, the keystone plant species in this community, the rattle *Rhinanthus alectorolophus* was directly affected by the sulfonylurea herbicide. The application resulted in a total decline of this hemiparasitic plant. *Rhinanthus* species represent a central food source for many bumble bee species and other FVIs (Kwak 1980; Hartley et al. 2015) and therefore indirect effects on these FVIs are likely. Additionally, a sublethal effect of the herbicide was the reduction of common buttercup flowers (*Ranunculus acris*): More than 100 insect species are known to visit this plant species (Weiner et al. 2011) and one specialist bee species (*Osmia florisomnis*) depends on its presence (Schmitz et al. 2013). These examples show that indirect effects of herbicides on FVIs that are so far not addressed in risk assessment might be even more severe than direct insecticide effects and therefore their risk is underestimated.

1.5 Conclusion

The EFSA bee guidance document (EFSA 2013) and NTA scientific opinion (EFSA 2015) incorporate substantial improvements of the current risk assessment process regarding FVIs. However, there still are deficits. One major problem is that bees and all other FVI species are subject to different risk assessment schemes. This complicates the implementation of an effective risk assessment process for the whole group of FVIs. Test species selection and representativeness is also an important issue. The honey bee as a test species might only be representative for other bee species due to its physiological and ecological properties. Even this is not trivial as shown for neonicotinoid insecticides (Cutler et al. 2014; Rundlöf et al. 2015). The preliminary selection of the additional test species *Bombus terrestris* and *Osmia bicornis* is disputable since there is uncertainty concerning their suitability as surrogate species. The existing information suggests that the honey bee is a highly sensitive species for tier I acute toxicity tests compared to *O. bicornis* and *B. terrestris*. However, more toxicity data are required for other pesticide classes than organophosphates to establish an appropriate bridging factor from honey bee acute studies (Arena & Sgolastra 2014; Uhl et al. 2016). This applies not only to acute toxicity endpoints but more so to measures of chronic toxicity since interspecific sensitivity differences are likely more pronounced after prolonged exposure especially for cumulatively toxic substances such as neonicotinoids (Heard et al. 2017). However, due to ecological differences, especially sociality, these two additional test species might be suitable to improve higher tier risk assessment (Cutler et al. 2014; Rundlöf et al. 2015). Introducing a lepidopteran surrogate test species for non-bee FVIs is a reasonable measure but this additional test species should be selected with great care. It is unclear if one species is enough to represent this ecologically diverse group since information on sensitivity of non-bee FVI species is scarce. Furthermore, exposure scenarios for such a diverse group as FVIs are hard to define since there is no comprehensive review of the exposure risk throughout FVI taxa and their life history up until now. The suggested caterpillar toxicity test includes oral exposure by residues on plant material. However there are other exposure pathways that need to be addressed for FVIs such as larval feeding on systemically contaminated plant parts (stem/leaves, pollen and nectar) or the consumption of exposed pollen and nectar or water by adults. EFSA proposals for bee semi-field and field studies are tackling well-known issues of these experiment types (EFSA 2013). However, appropriate test protocols need to be developed and validated. The upcoming NTA guidance document should specify test protocols for non-bee FVIs.

Overall, the recent EFSA revisions of bee (EFSA PPR Panel 2012; EFSA 2013) and NTA risk assessment (EFSA 2015) still do not sufficiently incorporate the ecological properties of FVI species. Furthermore, it is necessary to address indirect effects of herbicides on FVIs through food depletion. To achieve a protective risk assessment for FVIs, ecological characteristics of the different taxa of this group need to be addressed regarding exposure and effects. Furthermore, a set of effect measures needs to be defined, as well as, acceptable effect levels. When these prerequisites are met, more protective risk management measures can be formulated.

Table 1: Overview of relevant guidelines and their deficits (+ indicates an addition to the existing guidance).

Guidance document	Implemented	Test organisms	Effect assessment	Exposure assessment	Exposure scenarios	Deficits
Bees						
OEEP/EPPO (2010a, b)	Yes	A. mellifera	Tier 1: Acute contact/oral Higher tier: brood feeding, semi-field, field	Spray, nectar, pollen	In-field	Test organism not representative, insufficient exposure assessment
EFSA (2013)	No	+ *O. bicornis*, *B. terrestris*	Tier 1: + Chronic oral, chronic larvae Higher tier: + improved quality in semi-field, field protocols	+ Dust, guttation water, extrafloral nectaries, honeydew, surface waters	+ Off-field	Uncertainty about suitability of additional test species, SPG biodiversity is vague, missing or unvalidated semi-field and field test protocols, indirect effects not incorporated
Non-bee FVIs						
Candolfi et al. (2001)	Yes	*Aphidius rhopalosiphi, Typhlodromus pyri* & four proposed test species for higher tier assessment	Tier 1: Acute/chronic contact Higher tier: Extended lab, age residue, semi-field, field	Spray/residue	In-field, off-field	No FVI species, insufficient exposure assessment
EFSA (2015)	No	+ 1 lepidopteran species	Higher tier: + Landscape-scale assessment	+ Nectar, pollen, plant material, dust	No change	No guttation water in exposure assessment, uncertainty if additional test species is adequate surrogate for FVIs, SPG biodiversity is vague, lacking concrete guidance

2 Taxonomic groups of flower-visiting insects

Philipp Uhl, Carsten Brühl

2.1 Protection goals

FVIs are protected under several guidelines, declarations and regulations on a European and worldwide level. The UN "Convention on Biological Diversity" states the conservation of biological diversity and the sustainable use of the components of biological diversity as parts of its main objectives (United Nations 1992). Countries should implement national strategies to ensure these goals. In the EU the "Regulation No 1107/2009 of the European Parliament and the Council" was enforced to ensure that pesticides (plant protection products and active ingredients) are only placed on the market after it has been assessed that they are not harmful to human/animal health and the environment. It is further stated that pesticides shall have no unacceptable effects on non-target species and biodiversity in general (European Commission 2009). The European Food Safety Authority (EFSA) has released a "Scientific Opinion addressing the state of the science on risk assessment of plant protection products for non-target arthropods" where it defines protection goals using an ecosystem service approach. They defined five ecosystem services, four of which are fulfilled by FVI species (EFSA 2015):

- Biodiversity, genetic resources
- Education, inspiration and aesthetic value
- Regulation of arthropod pest populations (not applicable to FVIs)
- Food provision
- Pollination

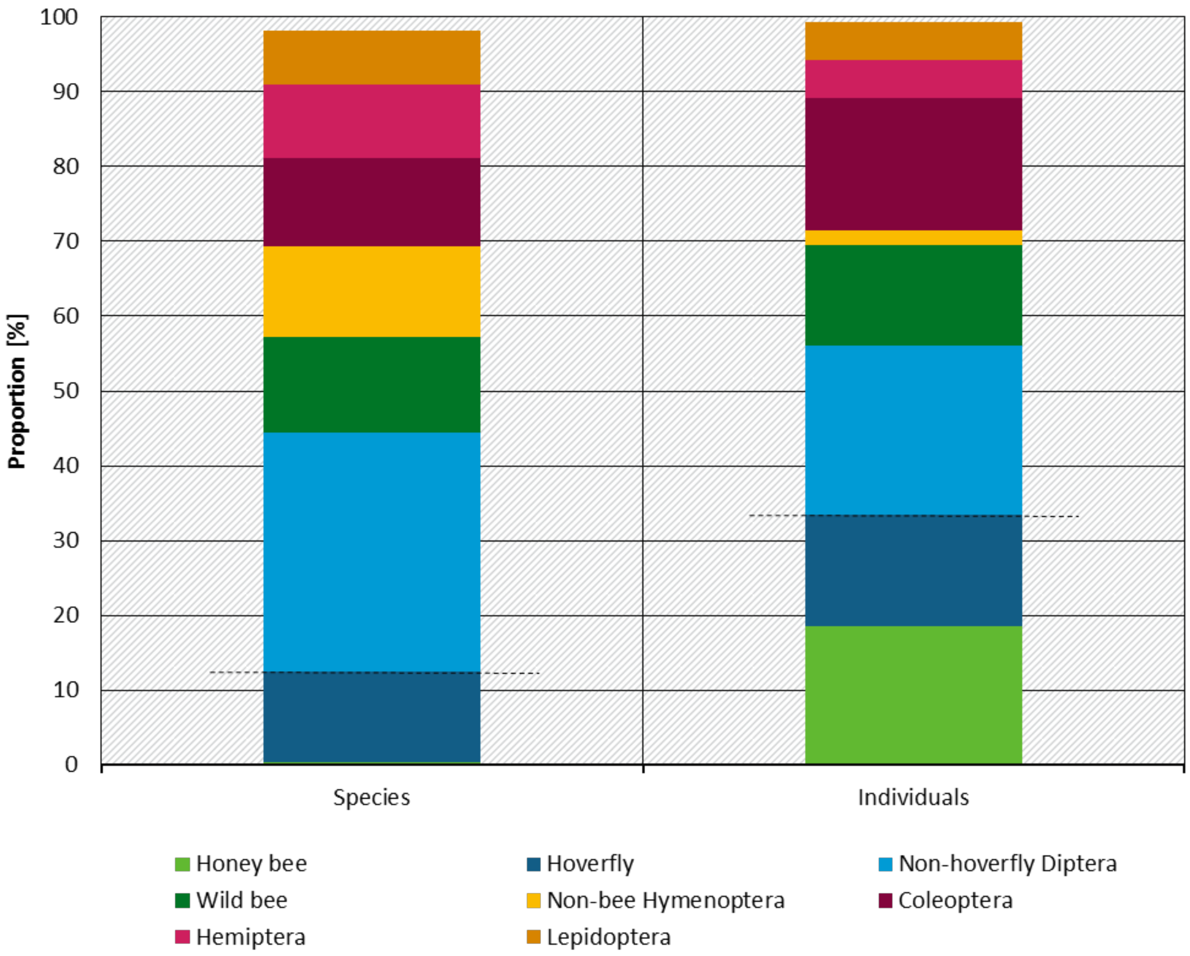
Regardless of the ongoing discussion concerning the ecosystem service concept (Mace et al. 2012) the protection of biodiversity is an integral part of this EFSA scientific opinion and most likely of the upcoming NTA guidance document. Furthermore, at the 13th meeting of the Conference of the Parties to the Convention on Biological Diversity (CBD) the "Cancun Declaration on Mainstreaming the Conservation and Sustainable Use of Biodiversity for Well-Being" was passed (United Nations 2016). Member states commit to take effective measures to counteract biodiversity loss. A guidance for the agricultural sector that lists actions to advance this goal includes the effective management and conservation of pollinators. Moreover, a "Coalition of the Willing on Pollinators" was created by 12 EU countries, including Germany, which commit to the following (Coalition of the Willing on Pollinators 2016):

- Take action to protect pollinators and their habitats by developing and implementing national pollinator strategies
- Share experience and lessons learnt in developing and implementing national pollinator strategies, especially knowledge on new approaches, innovations and best practices
- Reach out to seek collaboration with a broad spectrum of stakeholders – countries as well as businesses, NGO's, farmers, local communities
- Develop research on pollinator conservation
- Mutually support and collaborate with each other – and those parties that are willing to join the coalition

2.2 State of research

In the scientific literature the main groups of FVI species are often identified as bees, hover flies and lepidopterans (Winfree et al. 2011). However, more recent studies found that FVI communities in the agricultural landscape are far more diverse and the current definition of relevant groups seems incomplete. Grass et al. (2016) investigated flower visitations of insects in wildflower plantings situated in the central German agricultural landscape. They found that aside from bees and hover flies flowers were visited by a diverse community of other insect taxa. In fact, non-bee/non-hover fly insects made up half of the visiting individuals and 75% of FVI species (Figure 1). Furthermore, non-hover fly Diptera were by far the largest portion of visiting species. In contrast, butterflies only made up a small share of FVI abundance whereas flower visitation by beetles and non-hover fly Diptera in terms of individual numbers was comparable to that of honey bees. A meta-analysis by Orford et al. (2015) also emphasised the importance of non-syrphid flies as flower visitors. They analysed 71 plant-pollinator-visitation networks and 30 pollen-transport networks in agricultural, semi-natural and natural habitats from 11 projects. They found syrphids to exhibit slightly higher evenness in their flower interactions which results in more stable plant-pollinator communities. Pollen specialisation was, however, not different. Syrphids and non-syrphids essentially visited the same flowers. Furthermore, non-syrphids made up 82% of the dipteran abundance and 73% of dipteran species in farmland.

Figure 1: Contributions of different insect taxa to flower visitations in wildflower plantings.



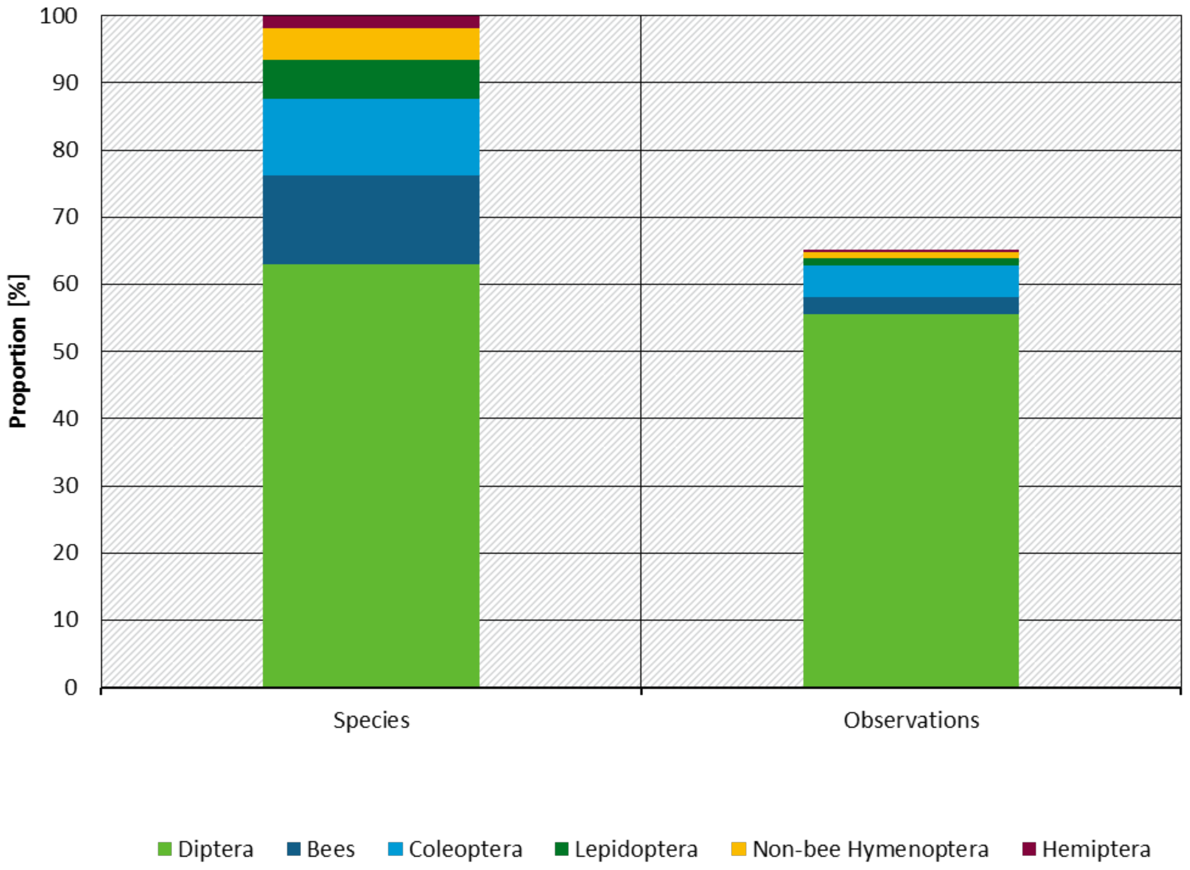
Footnote: The dashed line shows the cumulative fraction of honey bee and hover fly flower visits.

Source: own illustration. Data from Grass et al. (2016)

There is further evidence for the diversity of FVI communities. Common buttercup *Ranunculus acris* flower visitations were investigated in three German regions within the Biodiversity Exploratories. This is a large-scale, long-term project to assess the effects of land use on functional biodiversity

(Fischer et al. 2010). *R. acris* is a common plant of the agricultural landscape and widely distributed in Europe (Schmitz et al. 2013). Results show that flies were by far the most abundant group in species and individual observations (Figure 2). Furthermore, beetles showed similar species and individual numbers as bees which substantiates the findings of Grass et al. (2016) in a common wild plant.

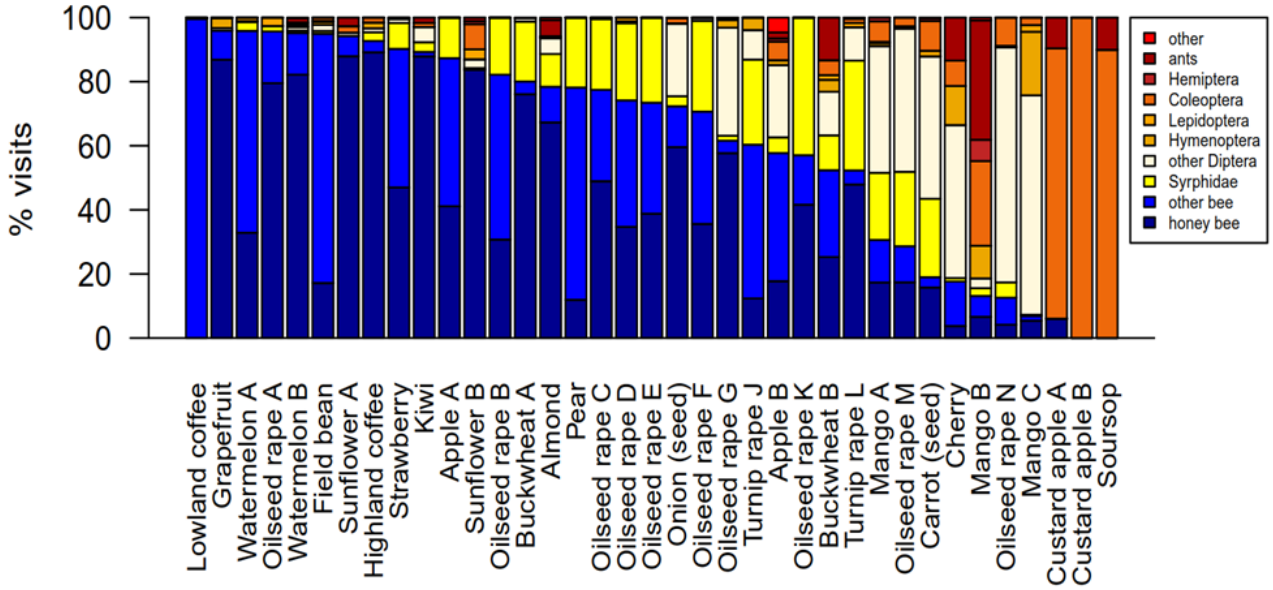
Figure 2: Flower visitations on *Ranunculus acris* by different insect taxa.



Source: own illustration. Data from Biodiversity Exploratories (2007-2008). Personal communication with Nico Blüthgen (2012)

In an extensive meta-analysis Rader et al. (2015) summarised the results of 39 field studies that investigated flower-visitation and pollination parameters in several crop systems from five continents. Overall, non-bee species accounted for 38% of flower visits (Figure 3). Distribution of visitation rates varied greatly between cropping systems and geographic locations. Custard apple crops in Australia and Brazil were not visited by any bee species whereas lowland coffee plants in India were exclusively visited by honey bees. The visitations by non-bees of oilseed rape as a typical European mass-flowering crop were also quite variable (between 5-80%) and varied even within countries (5-60%). Oilseed rape was visited by a substantial amount of syrphid and non-syrphid Diptera which was also true for cherry and apple.

Figure 3: Flower visitations by insect taxa in different crop systems.



Footnote: For some cropping systems there were multiple studies available that are distinguished by capital letters (e.g. "A", "B", "C").

Source: Rader et al. (2015)

2.3 Conclusion

With the current data we can define the relevant FVI groups in crops and their semi-natural surroundings as bees, flies (non-syrphids and syrphids), moths and butterflies and beetles. Proportions of species and individuals of the respective groups are rather variable in different crop systems (Rader et al. 2015) and semi-natural habitats (Grass et al. 2016). For non-bee Hymenoptera and Hemiptera there is not enough information available to make statements on their relevance as FVIs.

Despite clear identification of major FVI groups sufficient information to evaluate the risk of pesticides is only available for bees and Lepidoptera. There are eminent knowledge gaps regarding the ecological information for flies and beetles. Furthermore, there are little to no studies on the exposure of these taxa to pesticides and subsequent effects. This makes it difficult to draw conclusions about their risk and propose adequate protocols to assess it. Therefore, this report is focusing on bees and Lepidoptera but it will be noted if a section might apply to other FVI groups.

Open questions/Research opportunities

- The relevance of non-bee Hymenoptera and Hemiptera as flower-visiting insects needs to be evaluated by more studies on plant communities and their networks. This also applies to the less well-researched confirmed FVI groups moths, butterflies and beetles

3 Habitat requirements and ecological vulnerability

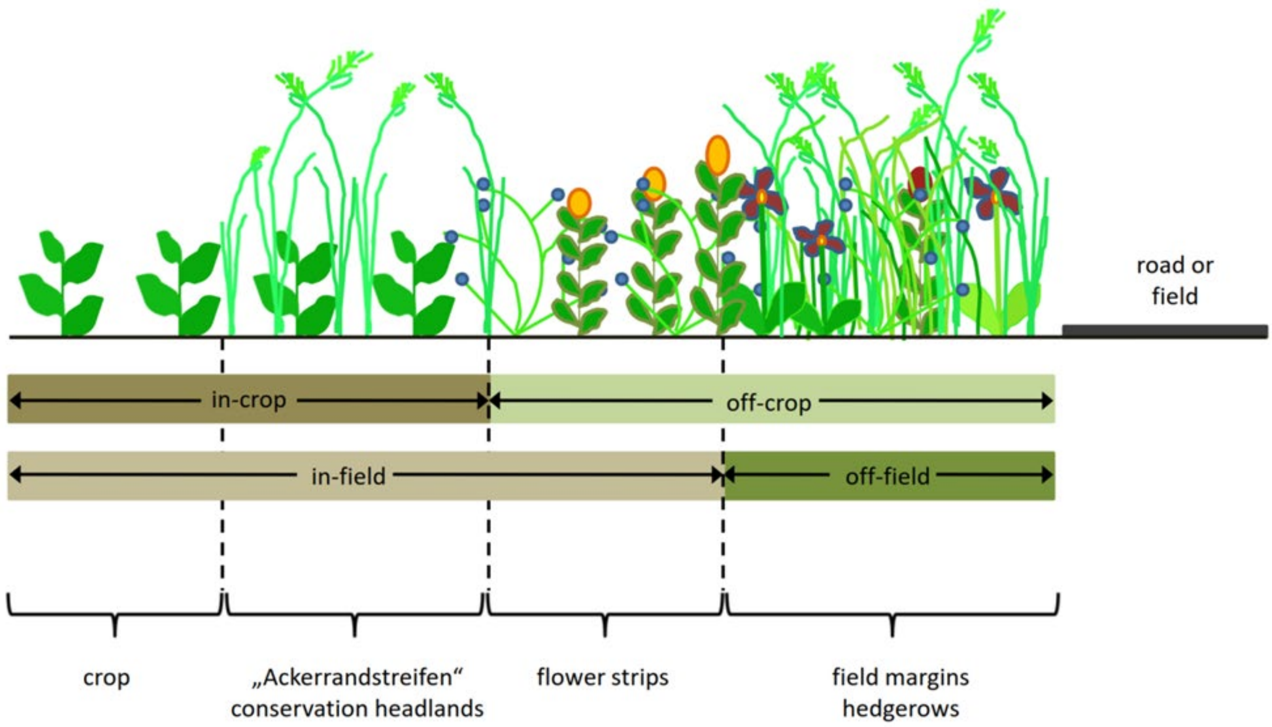
Philipp Uhl, Carsten Brühl

3.1 Habitat types

3.1.1 Introduction

The agricultural landscape provides diverse habitats for FVIs which can be categorised into three general types: in-crop/in-field (crop plantings), off-crop/in-field (managed flower/buffer strips) and off-crop/off-field (field edge structures) (Figure 4). These areas differ in many aspects such as structural composition, plant species inventory/diversity, spacial and temporal food resource availability, natural enemies or anthropogenic stress such as pesticide input.

Figure 4: Habitat types of FVIs in the agricultural landscape.



Source: Hahn et al. (2015a)

3.1.2 Crops

Cropping systems might differ in their suitability as FVI habitats due the attractiveness of the crop, structural differences and agricultural practices: The three main systems in Europe are arable crops, orchards and vineyards. EFSA classified the major European crops by their attractiveness to bees (EFSA 2013). The crops that were evaluated as possible food sources of bees are listed, assigned to a cultivation system, in Table 2. However, they noted that it is difficult to evaluate food source suitability of many crops since most studies are performed with honey bees and focus on major sources of pollen/nectar in their diet. Furthermore, EFSA included crops in their list which are grown for seed production and harvested before flowering. Their attractiveness to FVIs is therefore much smaller than in crops that are allowed to flower. Moreover, EFSA mentioned certain cereals that may attract FVIs through guttation (see 4.2 Relevant exposure scenarios for habitat matrices). Information on the full spectrum of crop resources in bee diets is scarce. Furthermore, many bee species are polylectic (food generalists) which means they will adapt their diet to the available floral resources in their foraging range (Michener 2007). Therefore, even less attractive crops might be used to collect pollen/nectar in times of food shortage or if they are very abundant. For other FVI species it is more difficult to confidently evaluate which crops might be suitable habitats since there is considerably less information available. It should be noted that in FVI groups such as Lepidoptera or beetles crops can be a habitat not only for adults but for their herbivore larval stages.

Table 2: European crops designated bee-attractive by EFSA.

Cultivation system	Crop	Comment
Arable crop	Alfalfa, asparagus, beans (*Phaseolus* ssp.), blueberries, broad beans/horse beans (*Vicia faba*), buckwheat, castor beans, chick peas, chillies and peppers, clover, cow peas, cranberries, cucumber and gherkin, currants, eggplant, gooseberries, peanuts, hemp, legumes for silage (e.g. *Lotus corniculatus*, *Lespedeza* spp., *Pueraria lobata*, *Sesbania* spp., *Onobrychis sativa*, *Hedysarum coronarium*), lentils, linseed, lupines, maize, melon, mustard, okra peas, peppermint, poppy, potatoes, pumpkins, squash and gourds, pyrethrum (*Chrysanthemum cinerariifolium*), oilseed rape, raspberries (and similar berries), safflower, cotton, serradella/birdsfoot (*Ornithopus sativus*), sesame, soybeans, spices (e.g. *Laurus nobilis*, *Anethum graveolens*, *Trigonella foenum-graecum*, *Crocus sativus*, *Thymus vulgaris*, *Curcuma longa*), strawberries, sugar beet, sunflower, tomatoes, vetches (*Vicia sativa*), viper's grass (*Scorzonera hispanica*), watermelons	
	Anise, badian fennel, corian, artichokes, cabbage and other brassica, carrots, cauliflower and broccoli, chicory, garlic, leeks and other alliaceous vegetables (e.g. *Allium porrum*, *A. schoenoprasum*), onions, tobacco, turnips	Harvested before flowering
	Barley, oats, rice, rye, rye grass for forage and silage (e.g. *Lolium multiflorum, L. perenne*), sorghum, triticale, wheat	Can attract FVIs by guttation
Orchard	Almonds, apples, apricots, avocados, bananas, carobs, cherries, chestnuts, coffee, dates, elder, figs, grapefruit, hazelnuts, kiwi fruit, lemons and limes, olives, oranges, peaches and nectarines, pears, persimmons, pistachios, plums and sloes, quinces, tangerine, mandarine and clementine, walnuts	
Vineyard	Grapes	

Adapted from EFSA (2013).

Even though it is difficult to assess the attractiveness of many crops there are some field cultures that are favourable for bees. Mass-flowering crops such as oilseed rape *Brassica napus* or sunflower *Helianthus annuus* are food sources for wild and managed bee species (Holzschuh et al. 2013; Requier et al. 2015; Sardiñas & Kremen 2015). These plants might not generally make up a great portion of bee species diet but due to their dominance in agriculture landscape and therefore abundant supply of floral resources they will be used as food sources (Coudrain et al. 2015; Requier et al. 2015; Sickel et al. 2015). Furthermore, even non-attractive crops might be FVI habitats if there is undergrowth of crop-associated wild plants either naturally occurring (e.g. cornflower or poppy species in cereal fields) (Storkey & Westbury 2007) or planted as integrated pest control measures (Balmer et al. 2014; Manandhar & Wright 2016). Moreover, even without any flowering plants present in-crop areas can still provide habitat functions for FVIs (e.g. nesting area for bees, flies and beetles or temporary refuge for all flying FVI life stages).

In absence of sufficient information and to exercise the precautionary principle it seems prudent to generally assume that all farmland areas represent FVI habitats if there is no contradicting evidence.

3.1.3 Field edge structures

There are multiple types of field-adjacent structures which differ in plant species inventory and habitat suitability for FVIs species. Marshall & Moonen (2002) defined a field margin as "the whole of the crop edge, any margin strip present and the semi-natural habitat associated with the boundary." They further state that this definition includes hedges and other kinds of barriers. In the context of this report this definition is generally applied to field-adjacent semi-natural habitats whereas the term field margin is specifically used for grassy or herbaceous vegetation strips adjoining to in-field areas (Table 3). Field edge structures are a major type of semi-natural habitat in intensely managed agricultural areas. They provide multiple habitat function for FVI species, e.g. refugia, feeding and breeding grounds and migration corridors (Marshall & Moonen 2002). More recent studies reinforce these statements: Denisow & Wrzesień (2015) performed transect walks along field margins in Polish farmlands. They concluded that field margins are important refugia for bee species and that they support pollinator biodiversity. Marshall et al. (2006) recorded higher abundances of bees at farmland sites where grassy field margins were sown. The plant communities of field edge structures are composed of wild plant species that contain woody flowering plants such as sloe (*Prunus spinosa*) in hedges and a high diversity of herbaceous flowering plants in field margins.

Table 3: Description of different field edge habitats.

Field edge structure	Description
Hedge	Shrubs and/or trees growing in one- to multi-rowed structures adjoining to agricultural sites
Copse	Small-scaled areas of bushes and trees (max. 2000 m²) within or next to agricultural sites showing a classification into tree- and shrub layer (height: more than 5 m)
Shrubbery	Small-scaled areas predominantly grown with shrubs (max. 2000 m²) within or next to agricultural sites (height: less than 5 m)
Forest edge	Transitional zone between agricultural sites and woods/forests (without clear cutting)
Field margin	Permanent vegetation strips (mostly grassy or herbaceous) adjoining to agricultural sites which were mown periodically
Woody structure margins	Grassy and/or herbaceous vegetation strips adjoining to woody structures

Table adapted from Hahn et al. (2015a).

3.1.4 Flower strips

Flower strips are used as agri-environmental management measures to improve nectar and pollen availability and thus enhance pollinator biodiversity amongst other things in several European countries. These strips are sown with seed mixtures of (regional) wild flowers within the field boundaries adjacent to the field edge (off-crop/in-field). These measures are particularly aimed at insect conservation with emphasis on sustaining pollinator populations to ensure crop pollination and favouring predacious beneficials to support biological pest control (Haaland et al. 2011; Hahn et al. 2015a; Tschumi et al. 2015). Therefore, they are designed to function as suitable habitats for FVI species which was demonstrated in numerous studies. Feltham et al. (2015) found a 25% higher crop flower visitation frequency by FVIs in strawberry fields with flower strips. In a review on crop yield enhancement by sustaining wild pollinators Garibaldi et al. (2014) concluded that flower strips can provide suitable food and nesting resources for bees and hover flies and enhance their species richness and abundance. They concluded that flower strips are one measure to transform agricultural landscapes into suitable habitats for FVI species.

3.1.5 Habitat compartments

FVI species use different parts of their habitats for specific functions at certain phases in their life cycle (Table 4). Depending on the ecological attributes of FVI species (see 3.2 Categories derived from ecological traits) relevant compartments of habitats vary in time and space.

Table 4: Parts of habitats used by FVI species

Compartment	Life stages	Function
Airspace	Adults	Food search (foraging), mate search, nest search
Flowers	Adults and nectar/pollen feeding larvae	Food collection (foraging), shelter, mating, nesting, nest material collection
Stem/leaves	Adults and herbivore larval stages	Food collection (foraging), shelter, mating, nesting, nest material collection
Soil	Adults and soil-dwelling larvae	Nesting, shelter
Waters reservoirs	Adults	Water collection / consumption

3.2 Categories derived from ecological traits

3.2.1 Bees

3.2.1.1 Introduction of group

Bees (Hymenoptera: Apoidea: Anthophila) are widely acknowledged as flower-visitors. They are a monophyletic group of about 20000 species worldwide (Nieto et al. 2014). In Europe there are 1965 species present from six families that can be functionally assigned to two groups: The long-tongued bees including Apidae and Megachilidae and short-tongued bees with Andrenidae, Colletidae, Halictidae and Mellittidae.

Wild and managed bee species are important pollinators of many crops and native flora (Klein et al. 2007; Ollerton et al. 2011). Aside from the well-known western honey bee *Apis mellifera* there are a multitude of ecologically quite variable species in Europe. Some species are eusocial, i.e. live in colonies or aggregations, but most species are solitary. Additionally, there are parasitic bee species that plant their eggs in brood cells of their host bee species or in some cases even subdue a colony to tend to their young (e.g. Psithyrus species) (Michener 2007; Goulson 2010). Adult bees collect pollen and nectar from flowering plants to feed themselves and supply provisions for their offspring. Females lay eggs on stored food and seal off brood cells (solitary species) or actively feed larval stages (eusocial species). Mature larvae spin cocoons in which pupation occurs. Adult females hatch from the cocoons, leave the nest, fly to flowers to collect food and mate with males to lay eggs again. There are several nesting strategies in bee species: Many species burrow into the soil to build their nest. Others occupy preexisting cavities in soil (e.g. mouseholes) or deadwood; some actively excavate deadwood. Furthermore, many species use collected materials, such as soil, resin or pieces of leaves and flower petals, to build parts of or their whole nest (Michener 2007). Concerning food sources there are generalist (polylectic) and specialist (oligolectic) bees. Oligolectic bees only forage on flowers of certain genera, in some cases even just of one specific plant (Westrich 1990; Michener 2007). The active flight period and length of flight in bees differs within species. Many species start mating and foraging flights in spring while others begin their adult phase not before summer and continue until autumn (Westrich 1990). Most species have only one brood per year (univoltine) whereas some lay eggs throughout the year (multivoltine). Voltinism varies with geography and climate in some species (Michener 2007). Daily activity peaks can be at midday but also in the morning and evening as shown for several bumble bee species (Thompson & Hunt 1999; Steen 2016). Bee species vary greatly in their foraging range, i.e. the distance they can cover to search for food resources: Measured distances range from hundreds of meters to ten or more kilometers (Zurbuchen et al. 2010).

Bees are integral parts of the insect fauna in agricultural landscapes. Since the majority of species are polylectic and therefore quite flexible in their food sources they can visit wild flowers as well as crops (Westrich 1990; Michener 2007). Studies have shown that mass-flowering crops such as oilseed rape *Brassica napus* or sunflower *Helianthus annuus* can be used by bees as food sources to sustain their populations (Holzschuh et al. 2013; Sardiñas & Kremen 2015). However, wild flowering plants seem to be essential to keep a diverse diet and subsidise bees between mass-flowering periods of crops (Requier et al. 2015; Sickel et al. 2015). Westrich (1990) lists *Brassica napus* (11% of German bee species), blackberry *Rubus fruticosa* (8%) and field mustard *Sinapis arvensis* (6%) as crops visited by the most German bee species. Furthermore, several crop-associated wild plants that grow in off-field areas are visited by many bee species: Common dandelion *Taraxacum officinale* (24% of German bee species), clover *Trifolium* ssp. (16%), thistle *Cirsium* ssp. (15%), knapweed *Centaurea* ssp. (15%), buttercup *Ranunculus* ssp. (9%), common yarrow *Achillea millefolium* (9%) and wild carrot *Daucus carota* subsp. *carota* (9%). Due to their ability to fly they can use off-field and in-field areas for foraging. Depending on the species' foraging range they might use from 3 ha to 300 km² (assuming maximum foraging distances of 100 m up to 10 km as radius). Therefore, bee habitats can include diverse compartments of semi-natural structures and farmland. A detailed analysis of bee communities in agricultural landscapes is a challenging task. Depending on the sampled habitats the number of species can range from 30 (e.g. intensively managed grassland with unfavourable climate) to over 100 (structure-rich habitats with high plant diversity and favourable climate) (Mohr et al. 1992). However, these data are quite variable since the number of individuals caught in Malaise traps can fluctuate greatly from 200 to 5000 between sampling dates (Mohr et al. 1992). In a study in apple orchards Russo et al. (2015) found further evidence that variation between years in sampled bee communities is high and might mask differences between different habitats.

In recent years it became more and more evident that bee species are in decline globally in occurrence, individual numbers and habitat ranges (Goulson et al. 2015; IPBES 2016). Substantial losses of domestic honey bee hives and simultaneous declines in wild bee diversity have been recorded since the last century in the EU and the USA (Committee on the Status of Pollinators in North America 2007; Potts et al. 2010, 2015; vanEngelsdorp & Meixner 2010; Goulson et al. 2015). This development is driven by multiple factors, partly of anthropogenic origin: Habitat loss and habitat fragmentation, pesticides and environmental pollution, decreasing resource diversity, invasive species, pathogens and climate change (Goulson et al. 2015). It is difficult to evaluate the threat status of European bee species in detail because of data scarcity. The IUCN European red list of bees reported a lack of sufficient information for more than half of all species. They listed 9.2% of bee species as threatened but estimated that due to uncertainty the actual proportion could lie between 4 and 60.7% (Nieto et al. 2014). The German red list might serve as an indicator since threat status of nearly all species was evaluated: 52.6% of 561 species have been classified as threatened (Westrich et al. 2011). In some European countries up to 65% of bee species are red-listed (Patiny et al. 2009).

3.2.1.2 Analysis of European bee traits

Within species communities there are patterns of ecological attributes. These traits (Table 5) can be used to group species that occupy similar ecological niches to predict their abundance and assess the susceptibility of their populations to environmental stress (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015).

Table 5: Ecological traits of European bee species and their implication for population susceptibility to environmental stress.

Trait	Explanation
Mobility	The foraging distance is correlated with bee size and determines how far bees can fly to collect food and nest building resources (Greenleaf et al. 2007; Michener 2007). Small species with low mobility have been shown to be vulnerable to intensive agriculture. Bigger, more mobile species are most likely more resistant since they can use more diverse foraging grounds in case of disturbance (de Palma et al. 2015).
Sociality	Social bee species colonies have higher foraging and reproductive capacity. This should allow them to better compensate against stressors compared to solitary bees (de Palma et al. 2015). However, due to the sheer amount of resources needed for a colony, these species might forage on a wider variety of plants which would increase chances of (multiple) pesticide exposure (Brittain & Potts 2011). This aspect will be discussed in section 5.1 . Parasitism is treated as a social strategy in this report since at least the larvae of parasitic species assume the strategy of the host and are therefore affected by stress in a similar way.
Nesting	Different strategies such as aboveground vs. belowground nesting or active nest excavation vs. renting may result in different vulnerabilities in bee populations. However, evidence is inconclusive which strategies are more robust (Williams et al. 2010; de Palma et al. 2015). Furthermore, pesticide exposure from different matrices might be dependent on the environmental compartments that bees nest in. This aspect will be discussed in section 5.1 .

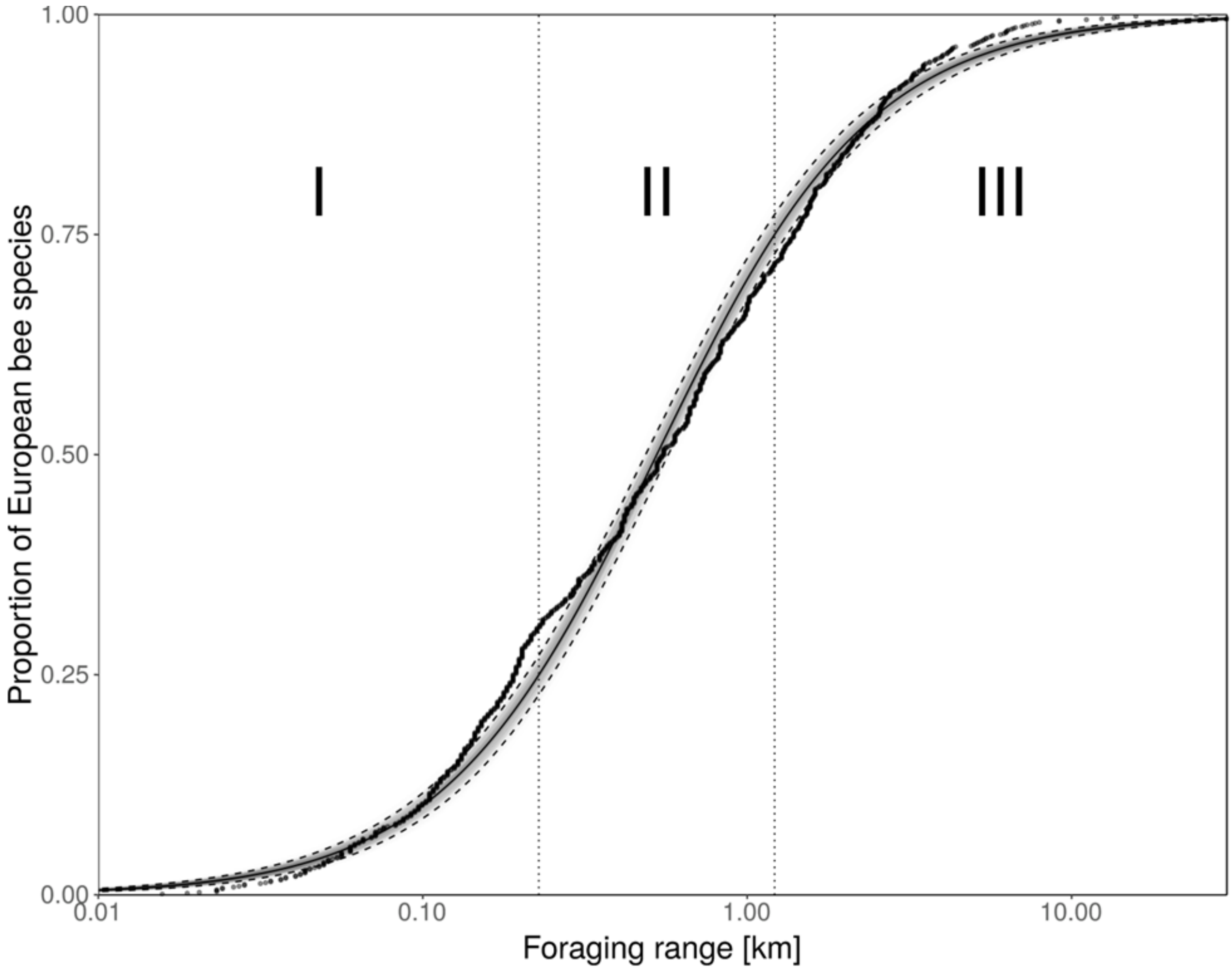
Trait	Explanation
Lecty	Dietary specialists (oligolectic species) react negatively to environmental stress (e.g. agricultural intensification, habitat loss) due to their limitation on few or just one food plant. Generalist, polylectic species can switch to alternative food plants (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015).
Flight season/duration	A short flight season corresponds with high sensitivity to stress events since the variety of plants that resources can be collected from and the time to do so is restricted. Species with longer flight seasons have more time to forage on additional plants (de Palma et al. 2015; Forrest et al. 2015).
Voltinism	Univoltine species might be vulnerable to changes in their environment in the time of reproduction whereas multivoltine species may be able to compensate due to two or more brood cycles within a year (Brittain & Potts 2011). However, this has not been clearly established.

Several traits were chosen to allocate bee species to categories of similar ecological vulnerability and choose representative species from these groups. To achieve this it is necessary to gather detailed information about these traits. A comprehensive database containing multiple trait data (e.g. foraging range, sociality, nesting strategy, flight months) for all European bee species (Roberts et al. unpublished; please see Acknowledgements for further information) was analysed to classify species by prevalent trait patterns.

Foraging ranges for bee species were estimated using the bees' intertegular distance (ITD; the width of the body measured between the wing bases) using the method from Greenleaf et al. (2007). Afterwards, foraging ranges for all European bee species were interpolated using the existing data (1003 out of 1936 species; Roberts et al. unpublished) and a log-logistic model. Bee species from this distribution were assigned to three groups (Figure 5). The 25% and 75% percentile of this distribution were chosen as cut-off values to represent the majority of species as well as minimum and maximum extreme groups:

- Category I: Low mobility. Foraging range from 15 m up to ca. 200 m (484 species)
- Category II: Medium mobility. Foraging range from ca. 200 m to ca. 1.2 km (968 species)
- Category III: High mobility. Foraging range ca. 1.2 to 24 km (484 species)

Figure 5: Foraging range of European bee species with interpolated categories.

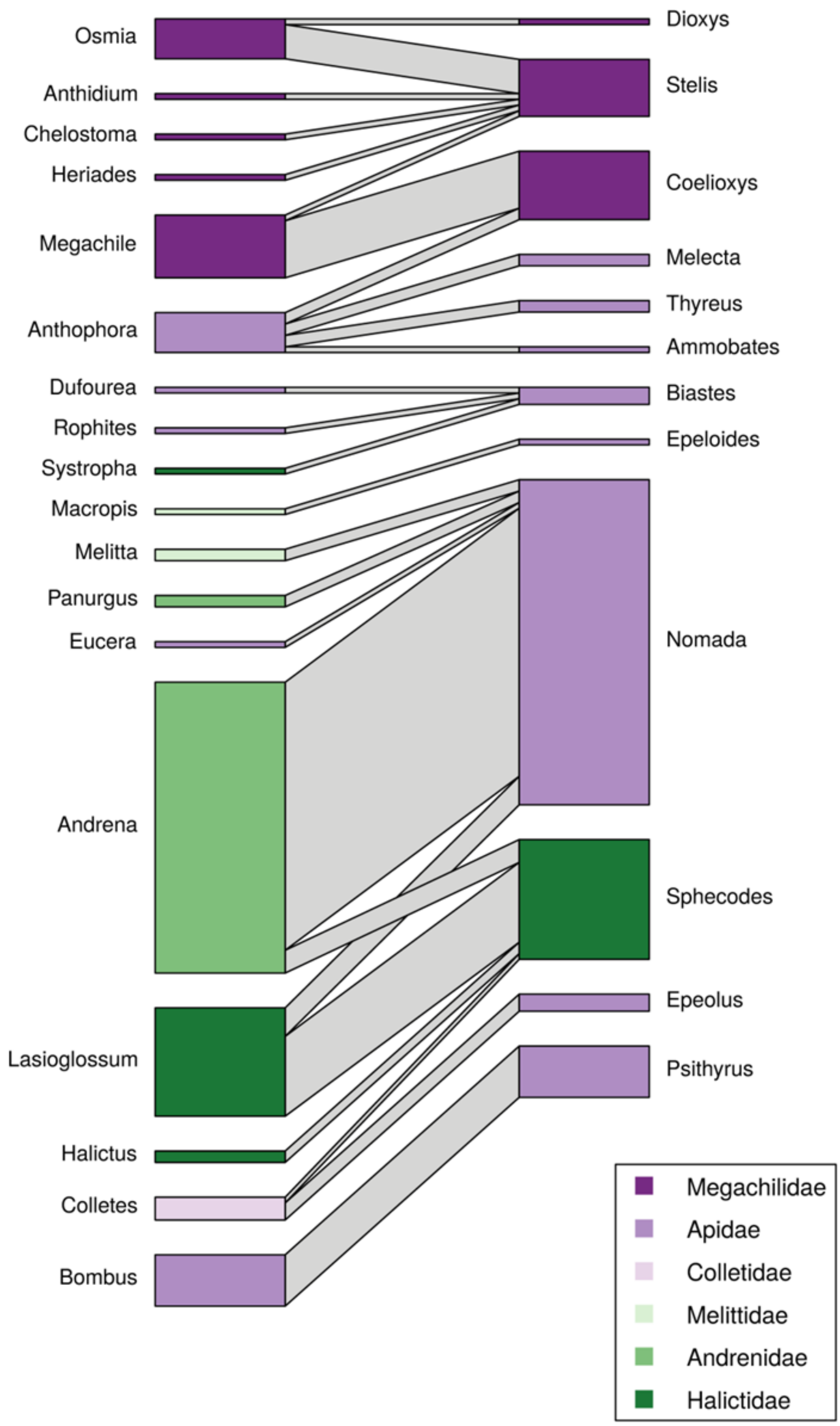


Footnote: I - low mobility, II - medium mobility, III - high mobility. Points show data for 1003 out of 1936 species. Species are assigned to three categories which are separated by dotted lines. The solid line describes a log-logistic model that was fitted to the data to interpolate to all European bee species. Dashed lines are 95% parametric bootstrap CIs and the grey area in-between shows all 1000 overlayed bootstrap samples.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

Social strategies can be classified in three predominant groups (data available for 1681 out of 1936 species; Roberts et al. unpublished): Solitary bees (73%), Parasites (21%) and primitively eusocial species (5%). Parasitic species are a comparatively large group among bee species which is dependent on their host populations. Therefore, social strategies and other traits (e.g. lecty of larvae) are copied from or adapted to the host. Unfortunately, there is no comprehensive dataset of parasite host associations of European bees. Therefore, parasites are evaluated as a separate group in this report whereas they could be integrated into other categories as shown exemplary for German bee genera (Figure 6).

Figure 6: Parasite host connections in German bee genera.



Footnote: Host genera are listed on the left which are connected to parasite genera on the right by grey lines. Size of the coloured box signifies the overall number of interactions of species from a certain genus whereas size of the connector lines represents the number of interacting species between a specific host and parasite genus. Colours signify taxonomic family.

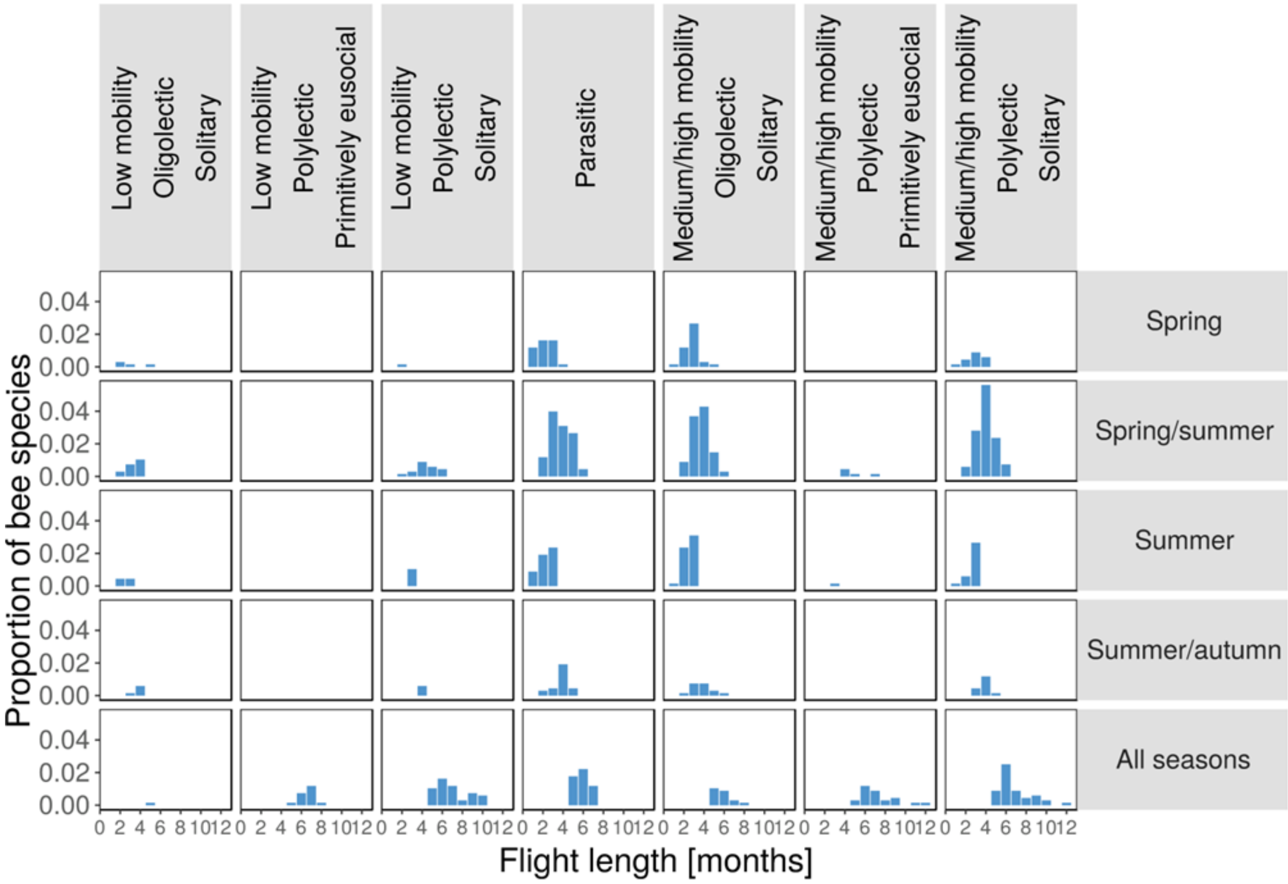
Source: own illustration. Data from Westrich (1990)

European bee species can be divided by their specialisation for food resources (data available for 922 out of 1936 species; Roberts et al. unpublished): 57% are polylectic and 40% oligolectic. When analysing voltinism we found that 93% of all species are univoltine and 5% bivoltine (data available for 1371 species). Since only a small portion of species has more than one brood cycle per year voltinism was disregarded for categorisation. However, this simplification might have to be revisited for regions with warmer climate where the proportion of multivoltine species might actually be substantially higher. Nesting strategies can be assigned to three main categories (data available for 1607 species; Roberts et al. unpublished): Excavating species that dig holes into the ground (64%), cleptoparasites (22%) and renters that use pre-existing cavities (9%). Since there is no conclusive evidence which effect this trait has on population sensitivity (Williams et al. 2010; de Palma et al. 2015) it was not used for the grouping of ecologically similar species.

European bee species were assigned to relevant categories considering the assessed ecological traits (Figure 7, Table 6). Of all possible combinations of traits several were merged, dropped or adapted: Foraging values of the bee database were calculated from the intertegular distance (ITD). The bases for these calculations are a regression formula derived from studies that analysed foraging distance (Greenleaf et al. 2007): Due to the methodology of these studies maximum foraging distances were often measured (e.g. translocation experiments, feeder training, bee dance interpretation). It is, however, highly doubtful that bees regularly fly such distances, especially in the case of highly mobile species (Zurbuchen et al. 2010). Due to this possible overestimation and the assumed higher vulnerability of species with limited mobility, medium and highly mobile species were merged (Figure 7). Parasitic bee species lay their eggs in brood cells of other species. Their larvae initially consume or kill the host egg and feed on brood provisions afterwards. In social host species they can also subdue host workers to tend to their young (Westrich 1990). Foraging ranges of the imagines are therefore not relevant since their exposure, e.g. during flower visitation, is negligible due to the fact that they do not forage for their brood. Therefore, the categories involving mobility or sociality were combined to one general parasitic group (Figure 7).

There are very few bee species that fly exclusively in autumn or in spring and autumn (only three species). Therefore, trait combinations including these species are disregarded because there are not quantitatively relevant. Oligolectic, primitively eusocial species are omitted for the same reason (only two species).

Figure 7: Distribution of European bee species between the main ecological trait categories.



Footnote: The percentage of bee species from these groups is shown in combination with flight length and flight season (678 out of 1936 species).

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

3.2.1.3 Focal species

After defining categories for all European bee species it is possible to assign focal species to these categories (Table 6). Focal species were chosen from the respective categories additionally considering a wide distribution in Europe and representative trait combination for their category with emphasis to their flight activity throughout the year. The resulting ecological profiles of derived focal species, that typically represent these categories, are listed below (Table 7-Table 13). Since trait data might not be available for similar proportions of all groups there might be some bias to the number of species allocated to each category.

Table 6: Overview of the ecological categories of European bee species derived from their traits.

Category	Ecological trait			Focal species	% of species
	Mobility	Lecty	Sociality		
A	low	oligolectic	solitary	*Andrena viridescens*	4.9
B	low	polylectic	primitively eusocial	*Lasioglossum malachurum*	1.7
C	low	polylectic	solitary	*Hylaeus communis*	8.3
D	low/medium/high	polylectic/ oligolectic	parasite	*Nomada striata*	38.0
E	medium/high	oligolectic	solitary	*Andrena proxima*	22.3
F	medium/high	polylectic	primitively eusocial	*Bombus terrestris*	3.4
G	medium/high	polylectic	solitary	*Osmia bicornis*	21.5

Data from European bee trait database (921 species out of 1936; Roberts et al. unpublished). Number of species per group is highly variable. Note that these values might be biased to a certain degree since trait data might not be available for similar proportions of all groups.

Table 7: Ecological profile of category A focal species *Andrena viridescens*.

Category	A – low mobility, oligolectic, solitary
Species	*Andrena viridescens*
Distribution	Temperate Europe
Habitat	Fertile meadows, orchard meadows, flood dams, calcareous grassland, vineyard fallows
Nesting	Excavator
Floral specialisation	*Veronica* ssp. (Scrophulariaceae)
Voltinism	Univoltine
Flight months	April-June
European IUCN red listings	2 countries

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

Table 8: Ecological profile of category B focal species *Lasioglossum malachurum*.

Category	B – low mobility, polylectic, primitively eusocial
Species	*Lasioglossum malachurum*
Distribution	Western palearctic
Habitat	Loess, loam and sand areas, clay and loam pits, forest edges, settlements
Nesting	Excavator, aggregations of nests
Social organisation	Three broods (two in northern area of distribution), queens overwinter and start first brood, males and queens in last brood
Voltinism	Univoltine
Flight months	April-October
European IUCN red listings	1 country

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

Table 9: Ecological profile of category C focal species *Hylaeus communis*.

Category	C – low mobility, polylectic, solitary
Species	*Hylaeus communis*
Distribution	All of Europe
Habitat	Ubiquitist, numerous habitats, e.g. forest edges, clearings, hedges, settlements, parks, ruderal areas, sand, loam and gravel pits, railroad embankments
Nesting	Excavator
Voltinism	Univoltine, partially bivoltine
Flight months	May-September
European IUCN red listings	none

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

Table 10: Ecological profile of category D focal species *Nomada striata*.

Category	D – low/medium/high mobility, parasitic, cleptoparasite
Species	*Nomada striata*
Distribution	Large parts of Europe
Host	Mainly *Andrena wilkella,* furthermore *A. ratisbonensis, A. gelriae, A. similis,* presumably also *A. intermedia, A. pandellei*
Voltinism	Univoltine
Flight months	April-July
European IUCN red listings	2 countries

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

Table 11: Ecological profile of category E focal species *Andrena proxima*.

Category	E – medium/high mobility, oligolectic, solitary
Species	*Andrena proxima*
Distribution	Nearly all of Europe
Habitat	Fertile meadows, flood dams, calcareous grassland, vineyard fallows, field margins, hedges, occasionally in settlements
Nesting	Excavator
Floral specialisation	*Anthriscus sylvestris*, *Daucus carota*, *Heracleum sphondylium*, *Aegopodium podagraria*, *Conium maculatum*, *Falcaria vulgaris*, *Chaerophyllum temulum* (Apiaceae)
Voltinism	Univoltine
Flight months	May-July
European IUCN red listings	1 country

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

Table 12: Ecological profile of category F focal species *Bombus terrestris*.

Category	F – medium/high mobility, polylectic, primitively eusocial
Species	*Bombus terrestris*
Distribution	All of Europe
Habitat	Ubiquitist, open landscape, settlements, gardens, parks
Nesting	Renter: Existing cavities
Social organisation	Two broods, queens overwinter and start first brood, males and queens in second brood
Voltinism	Univoltine
Flight months	All year
European IUCN red listings	none

Data from European bee trait database (Roberts et al. unpublished), Westrich (1990) and Michener (2007).

Table 13: Ecological profile of category G focal species *Osmia bicornis*.

Category	G – medium/high mobility, polylectic, solitary
Species	*Osmia bicornis*
Distribution	Central Europe and large parts of south and east Europe
Habitat	Forest edges, clearings, orchard meadows, hedges, vineyard fallows, settlements
Nesting	Renter: Existing cavities
Voltinism	Univoltine
Flight months	April-June
European IUCN red listings	none

Data from European bee trait database (Roberts et al. unpublished) and Westrich (1990).

3.2.2 Moths & butterflies

3.2.2.1 Introduction of group

Moths and butterflies (Lepidoptera) are a common and species-rich phytophagous insect group. Approximately 180000 Lepidoptera species have been described worldwide (Hamm & Wittmann 2009) and they account for approximately 10% of all known insect species (Willmer 2011). Although Lepidoptera are one of the most studied arthropod groups, the majority of Lepidoptera research has focused on diurnal butterflies (New 2004), which represent approximately 10% of the Lepidoptera species (Shields 1989). The remaining species are classified as moths and have often crepuscular and nocturnal lifestyles. For example, of the 3500 Lepidoptera species occurring in Germany (Karsholt & Razowski 1996), only 185 (5%) species are butterflies (Rhopalocera inclusive Hesperiidae, BfN 1998).

In the last decades, several studies have shown strong declines in moth and butterfly populations (Maes & Van Dyck 2001; Conrad et al. 2004, 2006b; Van Dyck et al. 2009; van Swaay et al. 2013). Two thirds of the 337 macro-moth species that were studied by Conrad et al. (2006b) with light traps in the UK have declined over a study period of 35 years. These species include many formerly common moths like *Arctia caj*a (Conrad et al. 2006a), *Eugnorisma glareosa*, and *Spilosoma lubricipeda* (Fox et al. 2006). Maes & Van Dyck (2001) found a decline in butterfly diversity and diversity hot spots in the agricultural intensified region of Flanders, Belgium. Van Dyck et al. (2009) analysed transect counts of butterflies in the Netherlands and could show substantial decreases in distribution and abundance of nine of 20 widespread species between 1992 and 2007. Evidence of negative pesticide effects on butterfly communities has been available even before the 1990s (Rands & Sotherton 1986). In general, especially two causes are discussed as main drivers for the declines in Lepidoptera: the effects of agricultural intensification, such as a loss of habitat and input of agrochemicals, and the consequences of climate change (Van Dyck et al. 2009; Fox 2012; van Swaay et al. 2013; Fox et al. 2014).

Many Lepidoptera species can be found in agricultural landscapes (see Annex I for a comprehensive list of scientific studies). The larvae (or caterpillars) of most lepidopteran species are herbivores and feed on plant material such as leaves, roots, flowers, seeds, or fruits (Scoble 1995). While some species are rather restricted in their caterpillar food spectrum and rely on one or a few host plants (monophagous or oligophagous caterpillars; e.g. European peacock *Aglais io*), others can consume a wide variety of plant species (polyphagous caterpillars; e.g. *Silver Y Autographa gamma*) (see Ebert (1994) for more information on host plants). As some Lepidoptera species also feed on crops during their caterpillar stage, they have been classified as agricultural pests such as caterpillars of the small white (*Pieris rapae*, Pieridae; feeding on cabbage), the cabbage moth (*Mamestra brassicae*, Noctuidae; various vegetables), the codling moth (*Cydia pomonella*, Tortricidae; apple), the European grapevine moth (*Lobesia botrana*, Tortricidae; grapes), or the vine moth (*Eupoecilia ambiguella*, Tortricidae; grapes). However, the majority of Lepidoptera species feed on non-crop plants (Ebert 1994; Scoble 1995).

In their adult stage, numerous Lepidoptera species feed on nectar and occasionally on pollen (Scoble 1995). The intake of nectar can improve longevity and reproduction (Mevi-Schütz & Erhardt 2005; Cahenzli & Erhardt 2012; von Arx et al. 2013). Hence, butterflies and moths are regularly observed flower visitors. Several studies have shown that Lepidoptera act as pollinators, but overall the knowledge on the role of Lepidoptera – and especially nocturnal moths – is limited (Hahn & Brühl 2016). In agricultural landscapes, research on flower visitors and pollinators often focus on crop plants and Lepidoptera have been observed visiting some crop plants (see Annex I). Nonetheless, in temperate regions, butterflies and moths probably play a minor role as crop pollinators (Hahn & Brühl 2016). However, since adult Lepidoptera visit a wide number of non-crop plant species as nectar sources, they might be of benefit to plant diversity due to pollination.

3.2.2.2 Analysis of European moths & butterfly traits

There are several studies on Lepidoptera (especially butterflies) focusing on the identification of ecological or functional characteristics to determine species' vulnerabilities to changes in environment and climate (Eskildsen et al. 2015; Aguirre-Gutierrez et al. 2016). The following ecological traits were considered to be of importance in several studies (see also Scalercio et al. 2012):

- Mobility of adults (Kuussaari et al. 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015)
- Habitat specialisation (Kuussaari et al. 2007; Eskildsen et al. 2015; Aguirre-Gutierrez et al. 2016)
- Host plant specialisation of the caterpillars (Franzen & Johannesson 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015; Aguirre-Gutierrez et al. 2016)
- Overwintering stage (Kuussaari et al. 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015)

Furthermore, other characteristics, such as the time and length of the adult flight period (Franzen & Johannesson 2007; Barbaro & van Halder 2009; Aguirre-Gutierrez et al. 2016) or the preferred growing conditions of host plants regarding nitrogen-input (eutrophication) (Kuussaari et al. 2007; Aguirre-Gutierrez et al. 2016) might be also of relevance.

In general, rare or declining species tend to be characterised by lower mobility, higher specialisation in their habitats and caterpillar host plants, and are overwintering as eggs or caterpillars (Kuussaari et al. 2007; Barbaro & van Halder 2009; Scalercio et al. 2012; Eskildsen et al. 2015). Hence, these traits can be suitable to characterise the ecological vulnerability of Lepidoptera species. For butterflies, many of these ecological and life-history traits are well studied (Ebert 1994; Settele et al. 2000). Probably due to their much higher species number and their often nocturnal activity phases, detailed information regarding ecological and life-history traits on moth species is rarely available. Nonetheless, some information can be found for macro-moths (see Ebert 1994; Pavlikova & Konvicka 2012).

Example species for ecological vulnerability

Highly vulnerable species:

The butterfly *Pseudophilotes baton* is categorised as threatened in Germany (National Red List). This species is described as rather sedentary, specialised in its caterpillar host plants (*Thymus pulegioides*, *T. serpyllum*) and habitat (e.g. xeric grassland or calcareous low-nutrient meadows), and it overwinters in the caterpillar stage (see Settele et al. 2000).

Least vulnerable species:

The butterfly *Pieris rapae* shows a high mobility as adult, overwinters as pupa, and is less specialised regarding caterpillar host plants (different Brassicaceae species) and habitat (e.g. fields, gardens, fallows, wetlands) (see Settele et al. 2000). This species is not threatened in Germany.

Eskildsen et al. (2015) used ordination methods and hierarchical clustering to distinguish Danish butterfly species on the basis of eight species traits into three functional groups (79 species classified):

- Category A: Sedentary habitat specialists overwintering in the egg stage
- Category B: Sedentary host plant specialists overwintering in the caterpillar stage
- Category C: Mobile generalist species overwintering as adults

Further assignment of butterfly species to ecological categories is not feasible at the moment since a comprehensive database of butterfly traits is not available (Eskildsen et al. (2015) only analysed a small dataset in terms of species number). These data would need to be collected for further analysis. Such an approach to generate functional groups using multiple ecological and life-history traits would probably also be applicable to moths where data is even more scarce.

3.2.2.3 Focal species

Since there are no definite ecological categories it is difficult to define focal species. Without proper reference such a selection will include a high level of uncertainty. In the absence of a general trait-based set of ecologically similar categories the above-mentioned categories by Eskildsen et al. (2015) may be used as provisional criteria to identify focal species.

Two butterfly species were used by Schuppener (2011) to assess effects und subsequent risk of genetically modified Bt maize: The small tortoiseshell *Aglais urticae* and the peacock *Aglais io*. Since they have been successfully used in ecotoxicity studies, both species can certainly be used as test species for pesticide risk assessment. They might also be suitable as focal species, since the larval foodplant, the nettle (*Urtica dioica*), is present in field margins and other off-crop structures of the agricultural landscape. *Aglais urticae* is a mobile, host plant specialist with mature overwintering stages (Table 14) whereas *Aglais io* is a sedentary species, host plant specialist and also overwinters in the adult stage (Table 15). However, these species do not completely fit the categories defined by Eskildsen et al. (2015) with their trait profiles (e.g. *Aglais urticae* could fit category C since it is a mobile species but on the other hand it is no food plant generalist). Furthermore, they are not representative of all ecological groups of Lepidoptera.

Table 14: Ecological profile of *Aglais urticae*.

Species	*Aglais urticae*
Distribution	All of Europe
Migratory	Yes (up to 100-150 km in migration flights)
Habitat	Hemerophile, occurs where larval food plant grows, hedges, parks, gardens, fallows, field margins, open habitats, adults are also found distant to suitable egg deposition areas
Nesting	Eggs are attached to bottom side of food plant leaves
Food plants (adults)	Wide spectrum, *Cirsium arvense*, *Centaurea jacea*, *Trifolium pratense*, *Eupatorium cannabinum*, *Buddleja* ssp., *Dianthus barbatus*, *Aster* ssp., *Tussilago farfara*, *Daphne mezereum*, *Salix caprea*
Food plants (larvae)	*Urtica dioica*, occasionally *Urtica urens* & *Humulus lupulus*
Voltinism	Univoltine or multivoltine (2 to 3 generations per year)
Flight months	March-April, June-July, (August-September)
Overwintering stage	Adult

Ecological information collated by Schuppener (2011).

Table 15: Ecological profile of *Aglais io*.

Species	*Aglais io*
Distribution	All of Europe (except far north and parts of Iberian Peninsula and Greece)
Migratory	No (migratory flights are rarely observed)
Habitat	Open habitats, forest edges, sunny clearings, humid/wet meadows, fallows, gardens, parks
Nesting	Eggs are attached to bottom side of food plant leaves
Food plants (adults)	Violet flowering plants, *Tussilago* ssp., *Salix* ssp., *Daphne* ssp., *Prunus spinosa*, *Prunus* ssp., *Taraxacum officinale*, *Eupatorium*, *Cirsium palustre*, *Cirsium arvense*, *Buddleja* ssp., overrife fruit in autumn
Food plants (larvae)	*Urtica dioica*, possibly other *Urtica* species and *Humulus lupulus*
Voltinism	Univoltine to multivoltine (2 to rarely 3 generations per year)
Flight months	May-July, August-September
Overwintering stage	Adult

Ecological information collated by Schuppener (2011).

There are a few other butterfly species that might be chosen at least as test species. Two species that have been investigated in several studies are the large white *Pieris brassicae* and the small white *Pieris rapae* which are regularly encountered in the agricultural landscape and are also pests for certain crops (Sinha et al. 1990; Davis et al. 1991b; Cilgi & Jepson 1995).

There are insufficient data to propose focal species for moths. However, there are several moth species that declined in the last decades (e.g. Table 16). It might be prudent to choose moth focal species from this group when minimal data requirements for trait analysis are met. It is essential to collate a comprehensive database of ecological information for these underrepresented lepidopteran species. However, it is possible to suggest test species. The corn earworm *Helicoverpa zea* has been established as a test organism in laboratory and field efficacy studies of Bt cotton (Jackson et al. 2003, 2004; Ali et al. 2006). Further species that have already been subject to ecotoxicological studies and could be established as test species are the cabbage moth *Mamestra brassicae* (Seljasen & Meadow 2006; Hahn et al. 2014), the diamondback moth *Plutella xylostella* (Kumar & Chapman 1984; Han et al. 2012) or the codling moth *Cydia pomonella* (Knight & Flexner 2007).

Table 16: Larger Britain moth species which populations declined by 75% or more between 1968 and 2007.

Species		Population trend [%]	Species		Population trend [%]
V-moth	*Macaria wauaria*	-99	Brindled Beauty	*Lycia hirtaria*	-87
Garden Dart	*Euxoa nigricans*	-98	Small Square-spot	*Diarsia rubi*	-87
Double Dart	*Graphiphora augur*	-98	September Thorn	*Ennomos erosaria*	-87
Dusky Thorn	*Ennomos fuscantaria*	-98	Sprawler	*Asteroscopus sphinx*	-87
Hedge Rustic	*Tholera cespitis*	-97	Rosy Rustic	*Hydraecia micacea*	-86
Figure of Eight	*Diloba caeruleocephala*	-96	Sallow	*Xanthia icteritia*	-85
Spinach	*Eulithis mellinata*	-96	Latticed Heath	*Chiasmia clathrata*	-85
Dark Spinach	*Pelurga comitata*	-96	August Thorn	*Ennomos quercinaria*	-85
Heath Rustic	*Xestia agathina*	-95	Oblique Carpet	*Orthonama vittata*	-85
Anomalous	*Stilbia anomala*	-94	Mouse Moth	*Amphipyra tragopogonis*	-85
Dusky-lemon Sallow	*Xanthia gilvago*	-94	Broom Moth	*Melanchra pisi*	-84
White-line Dart	*Euxoa tritici*	-94	Mottled Rustic	*Caradrina morpheus*	-84
Flounced Chestnut	*Agrochola helvola*	-94	Large Wainscot	*Rhizedra lutosa*	-83
Brindled Ochre	*Dasypolia templi*	-94	Brown-spot Pinion	*Agrochola litura*	-82
Autumnal Rustic	*Eugnorisma glareosa*	-94	Minor Shoulder-knot	*Brachylomia viminalis*	-82
Rosy Minor	*Mesoligia literosa*	-93	Green-brindled Crescent	*Allophyes oxyacanthae*	-81
Lackey	*Malacosoma neustria*	-93	Deep-brown/Northern Deep-brown Dart agg.	*Aporophyla lutulenta /luneburgensis*	-81
Grass Rivulet	*Perizoma albulata*	-93	Lead/July Belle agg.	*Scotopteryx mucronata/lurid ata*	-81
Large Nutmeg	*Apamea anceps*	-93	Small Autumnal Moth	*Epirrita filigrammaria*	-81
Beaded Chestnut	*Agrochola lychnidis*	-93	Grey Chi	*Antitype chi*	-80
Garden Tiger	*Arctia caja*	-92	Buff Arches	*Habrosyne pyritoides*	-80
Haworth's	*Celaena*	-92	Galium Carpet	*Epirrhoe galiata*	-79

Species		Population trend [%]	Species		Population trend [%]
Minor	*haworthii*				
Dark-barred Twin-spot Carpet	*Xanthorhoe ferrugata*	-91	Rustic	*Hoplodrina blanda*	-78
Dot Moth	*Melanchra persicariae*	-91	Oak Hook-tip	*Watsonalla binaria*	-78
Grey Mountain Carpet	*Entephria caesiata*	-91	Gothic	*Naenia typica*	-76
Broom-tip	*Chesias rufata*	-90	Heart and Dart	*Agrotis exclamationis*	-76
Pale Eggar	*Trichiura crataegi*	-90	Neglected Rustic	*Xestia castanea*	-76
Feathered Gothic	*Tholera decimalis*	-89	Knot Grass	*Acronicta rumicis*	-75
Oak Lutestring	*Cymatophorima diluta*	-88	Black Rustic	*Aporophyla nigra*	-75
Red Carpet	*Xanthorhoe decoloraria*	-88	Garden Carpet	*Xanthorhoe fluctuata*	-75
Pretty Chalk Carpet	*Melanthia procellata*	-88			

Table adapted from Butterfly Conservation (2013).

3.3 Definition of ecologically vulnerable groups

3.3.1 Bees

Depending on the breadth of the ecological niche the seven categories of bee species identified in section 3.2.1.2 Analysis of European bee traits can be evaluated by the ecological vulnerability of their populations (see Table 5). The narrower the niche the higher the sensitivity to external stressors (Williams et al. 2010; de Palma et al. 2015; Forrest et al. 2015). The three major traits identified for categorisation (mobility, sociality, lecty) of bee species plus flight season/length were used for the qualitative evaluation process. Depending on the number of criteria of these attributes that where met groups were defined as:

- Least vulnerable: Zero vulnerable trait states
- Medium vulnerable: One vulnerable trait state
- Highly vulnerable: Two or more vulnerable trait states

Species from **category A** are not very mobile which makes them susceptible to environmental change because they cannot migrate away from stressors (1). They depend on very specific food resources (2). Since they are also solitary they cannot buffer losses of individuals as social bees (3). Furthermore, their flight duration is comparatively short which means that stress events may not be compensated (4). This is further emphasised by the fact that nearly no species from this category flies from spring to autumn (Table 17).

Four criteria met: Highly vulnerable

Category B species are also rather immobile (1). Other than category A species they are not specialised in their flower preference and benefit from their social organisation to withstand stress. All species from this category fly from spring to autumn for five to eight month.

One criterion met: Medium vulnerable

In comparison to category A species of **category C** are not dependent to specific flower resources. They are a more heterogeneous group in terms of their flight length and season. Nonetheless, they are only slightly mobile (1) and solitary species (2) which makes them susceptible to environmental change.

Two criteria met: Highly vulnerable

Category D species are parasites and therefore rely heavily on host species abundance. Other than that the group is quite heterogeneous. It is necessary to further analyse connections to host species to make a conclusive statement.

Due to data scarcity and subsequent uncertainty: Medium vulnerable

Category E species need specific food plants (1) but are mobile enough to avoid inappropriate environmental settings by migration. Furthermore, they are solitary species (2).

Two criteria met: Highly vulnerable

Categories F species can fly greater distances and are polylectic. Furthermore, these species can compensate individual losses since they are social bees.

Zero criteria met: Least vulnerable

In contrast to category F species of **category G** are solitary bees (1). Furthermore, they are polylectic which should make even species with short flight length more resistant to environmental stress.

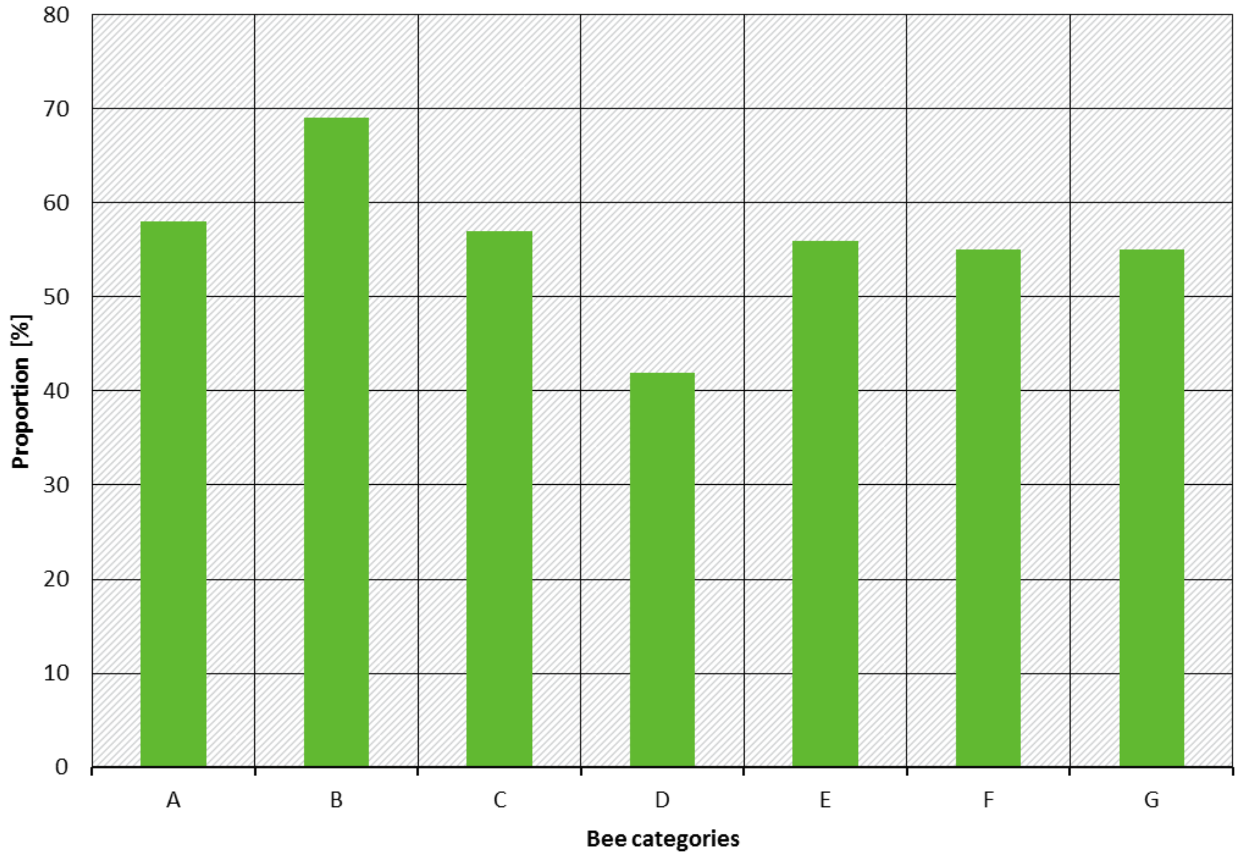
One criterion met: Medium vulnerable

Table 17: Bee categories and their ecological vulnerability.

Category	Ecological trait repertoire	Focal species	Vulnerability group
A	low mobility, oligolectic, solitary, short flight length, flight in spring and summer, not from spring to autumn	*Andrena viridescens*	highly
B	low mobility, polylectic, primitively eusocial, medium flight length, fly from spring to autumn	*Lasioglossum malachurum*	medium
C	low mobility, polylectic, solitary, short to long flight length, mostly fly from spring to summer or spring to autumn	*Hylaeus communis*	highly
D	parasitic, short to medium flight length	*Nomada striata*	medium
E	medium/high mobility, oligolectic, solitary, short to medium flight length	*Andrena proxima*	highly
F	medium/high mobility, polylectic, primitively eusocial, medium to long flight length, nearly all fly from spring to autumn	*Bombus terrestris*	least
G	medium/high mobility, polylectic, solitary, short to long flight length,	*Osmia bicornis*	medium

These vulnerability groups should be confirmed by actual population records. Only mid- to long-term monitoring can show if populations of designated vulnerable categories are actually decreasing and therefore susceptible to environmental change. Unfortunately, these data are scarce for most European bee species. There are data for the IUCN red list threat status in multiple European countries which are still fragmentary (Nieto et al. 2014). An analysis of the amount of species that are red-listed in at least one European country shows no clear picture (Figure 8). Nearly all categories have the same amount of threatened species (around 60%). Category B includes more red-listed species than the rest and category D less. However, due to the patchy data no reliable conclusions can be drawn. Therefore, a pending need for a Europe-wide population monitoring of bees was recognised (Nieto et al. 2014).

Figure 8: Proportion of red-listed species in at least one European country per category.



Footnote: Red list data of 21 European countries (921 out of 1936 species). Categories are defined in Table 6.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

3.3.2 Moths & butterflies

Due to data scarcity it was not possible to assign lepidopteran species to definite categories. Therefore, a definition of ecologically vulnerable groups is not feasible. From the available literature several traits were identified that are relevant to assess population vulnerability:

- Mobility of adults (Kuussaari et al. 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015)
- Habitat specialisation (Kuussaari et al. 2007; Eskildsen et al. 2015; Aguirre-Gutierrez et al. 2016)
- Host plant specialisation of the caterpillars (Franzen & Johannesson 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015; Aguirre-Gutierrez et al. 2016)
- Overwintering stage (Kuussaari et al. 2007; Barbaro & van Halder 2009; Eskildsen et al. 2015)

- Time and length of adult flight (Franzen & Johannesson 2007; Barbaro & van Halder 2009; Aguirre-Gutierrez et al. 2016)
- Preferred growing conditions of host plants regarding nitrogen-input (eutrophication) (Kuussaari et al. 2007; Aguirre-Gutierrez et al. 2016)

Using these ecological attributes it will be possible to comprehensively evaluate vulnerable traits combinations in European lepidopteran species if necessary data are collated.

3.4 Conclusion

Within the agricultural landscape crops as well as field edge structures and flower strips are habitats of FVI species. Non-attractive crop fields might still be FVI habitats due to weedy undergrowth consisting of flowering wild plant species. There are multiple habitat compartments that these species use throughout their life cycle (e.g. airspace, flower, stem/leaves, soil, water reservoirs). Ecological attributes determine the vulnerability of FVI populations towards stressors and there already is a comprehensive database for many traits of bee species to derive ecologically similar groups and choose focal species. These groups can also be evaluated regarding their vulnerability. However, theoretically vulnerable groups should not be mistaken for actually threatened groups. Actual threat still needs to be determined by population monitoring. For lepidopterans some traits have been identified that distinguish ecologically similar groups and might give information about their populations' vulnerability but there is not enough data to conclusively define similar groups or choose focal species. There is a need for further studies on the ecological attributes of moths, butterflies and other non-bee FVI species.

Open questions/Research opportunities

- The database on FVI habitats is small and needs to be expanded.
- Traits data need to be collected for lepidopteran and FVIs other than bee species to identify ecologically vulnerable groups.
- Increased population monitoring of FVI species is needed to confirm and assess vulnerable groups.

4 Pesticide exposure of habitats

Philipp Uhl, Carsten Brühl

4.1 Exposure processes

There are several transport processes by which pesticides can reach FVI habitats. These can generally be assigned to two distinct groups:

4.1.1 Primary processes

- These processes designate the intended application of pesticides to crops (in-field habitat):
- Spray application

Pesticides are administered diluted in a water-based mist over the crop. Usually pesticides with non-systemic properties are applied with this method. However, some formulations of systemic pesticides are also applied by spraying.

- Systemic application
  - Solid application (Seed treatment / granular application)

The other common method is to apply solid products of systemic pesticides. This can be done via seed-treatment where seeds are coated with a formulation of the pesticide which is taken up systemically by the plant and distributed within its tissues (Nuyttens et al. 2013). Furthermore, applications of granular formulations to the soil are also common.

  - Stem application

Another method relevant for permanent crops and less frequently used at least in Europe is stem applications. In this procedure a hole is drilled into the stem of a tree and the pesticide in applied directly into the xylem (Helson et al. 2001). This system is also feasible for viticulture (Düker & Kubiak 2015). However, it is not well-established in Europe, yet, and therefore not considered in current pesticide risk assessment.

  - Irrigation

Crops can be applied with pesticides using irrigation systems (e.g. Miorini et al. 2017). This application technique is also not yet considered in risk assessment since it is an uncommon practice in Europe.

4.1.2 Secondary processes

These processes describe the unintentional redirection of a fraction of the applied pesticide amount to in-field (e.g. crop, flower/buffer strips) and off-field habitats (e.g. field margins):

- Spray drift

During spray application a portion of the spray mist is dislocated away from the intended area by air flow into adjacent non-target habitats. This drift deposition decreases with increasing distance to the in-field area (Ganzelmeier et al. 1995; Rautmann et al. 2001). The amount of drift is higher in 3-D applications in fruit orchards and vineyards and lower in 2-D applications of arable crops.

- Field edge overspray

When applying pesticides by spraying the field edge is usually exceeded by the outermost spray cone therefore applying the adjacent non-target habitat with 50% application rate. This is done to ensure a 100% application rate at the field edge by overlap of the last two spray cones (Schmitz et al. 2013).

Therefore a part of the field margin (50 cm to 1 m off-crop habitat depending on spray boom height) is receiving an overspray of 50% application rate.

- Dust dispersion/drift

Pesticide-treated seed lose a differing amount of dressing particles in the sowing process that are released as dust. This is the result of the dressing process, storage, handling, movement and the actual sowing. These dust emissions can dislocate into non-target habitats. Dust deposition decreases with increasing distance to the in-field area (Nuyttens et al. 2013).

- Run-off

Pesticides are washed off the in-field area during (heavy) rain events and subsequently flushed into adjacent non-target habitats (Walker 2001).

4.2 Relevant exposure scenarios for habitat matrices

4.2.1 Introduction

A wide array of habitat compartments are reached vie primary and secondary processes (i.e. intended application or unintended redirection). These processes can also lead to combined exposure (Figure 9). In this section exposure scenarios for FVI habitat matrices are described. Scientific evidence of field exposure of these matrices is collated and discussed in section 5.2 Evidence of exposure by residue levels of pesticides. Exposure scenarios are summarised in Table 18 & Table 19.

Table 18: Potential exposure of FVI habitat matrices after pesticide spray applications in crop and field edge areas.

Matrix	In-crop	Off-crop	
		In-field	Off-field
Airspace	Overspray	Field edge overspray, spray drift	Field edge overspray, spray drift
Pollen/nectar			
Extrafloral nectaries			
Honeydew			
Guttation water			
Stem/leaves			
Soil		Field edge overspray, run-off	Field edge overspray, run-off
Water bodies		Field edge overspray, spray drift, run-off	Field edge overspray, spray drift, run-off

Table 19: Potential exposure of FVI habitat matrices after systemic pesticide applications in crop and field edge areas.

Matrix	In-crop	Off-crop	
	In-field		Off-field
Airspace	Dust dispersion	Dust dispersion	Dust dispersion
Pollen/nectar	Systemic load	Run-off, systemic load	Dust dispersion, run-off, systemic load
Extrafloral nectaries			
Honeydew			
Guttation water			
Stem/leaves			
Soil	Dust dispersion, systemic load	Dust dispersion, run-off	Dust dispersion, run-off
Water bodies	Dust dispersion		

4.2.2 Airspace

Spray application and subsequent volatilization of pesticides leads to temporary exposure of the airspace above a crop. Spray mist can drift into the airspace of off-crop areas (Ganzelmeier et al. 1995; Rautmann et al. 2001). Pesticide dust from systemic seed-treatment or granules application is also mobile enough to reach to off-crop habitats (Nuyttens et al. 2013).

4.2.3 Plant body (Pollen/nectar, stem/leaves, guttation water, extrafloral nectaries, honey dew)

Flowers of crops and weedy undergrowth are oversprayed in-crop and therefore pollen and nectar are exposed to pesticides. Due to field edge overspray and spray drift off-crop non-target plants are exposed as well. Furthermore, pesticide dust drift can lead to deposits on in-crop and off-crop flowers (Botías et al. 2016). Moreover, dust drift and run-off can result in systemic pesticide loads in off-crop soils which are taken up by non-target plants and are deposited in pollen and nectar. The same pathways are relevant for stem and leaves of crops and weeds/non-target plants in in-crop and off-crop habitats as well as guttation water, extrafloral nectaries and honeydew. Guttation is a process were water droplets are exuded from secretory tissues of many plant species and can be a water source for foraging FVIs (see below) (Bonmatin et al. 2015). Extrafloral nectaries are produced by a number of flowering plants and are a food source for many FVIs (Weber et al. 2015). Furthermore, aphids excrete honeydew which is utilised by honey bees, bumble bees and solitary bees amongst FVI species (Konrad et al. 2009).

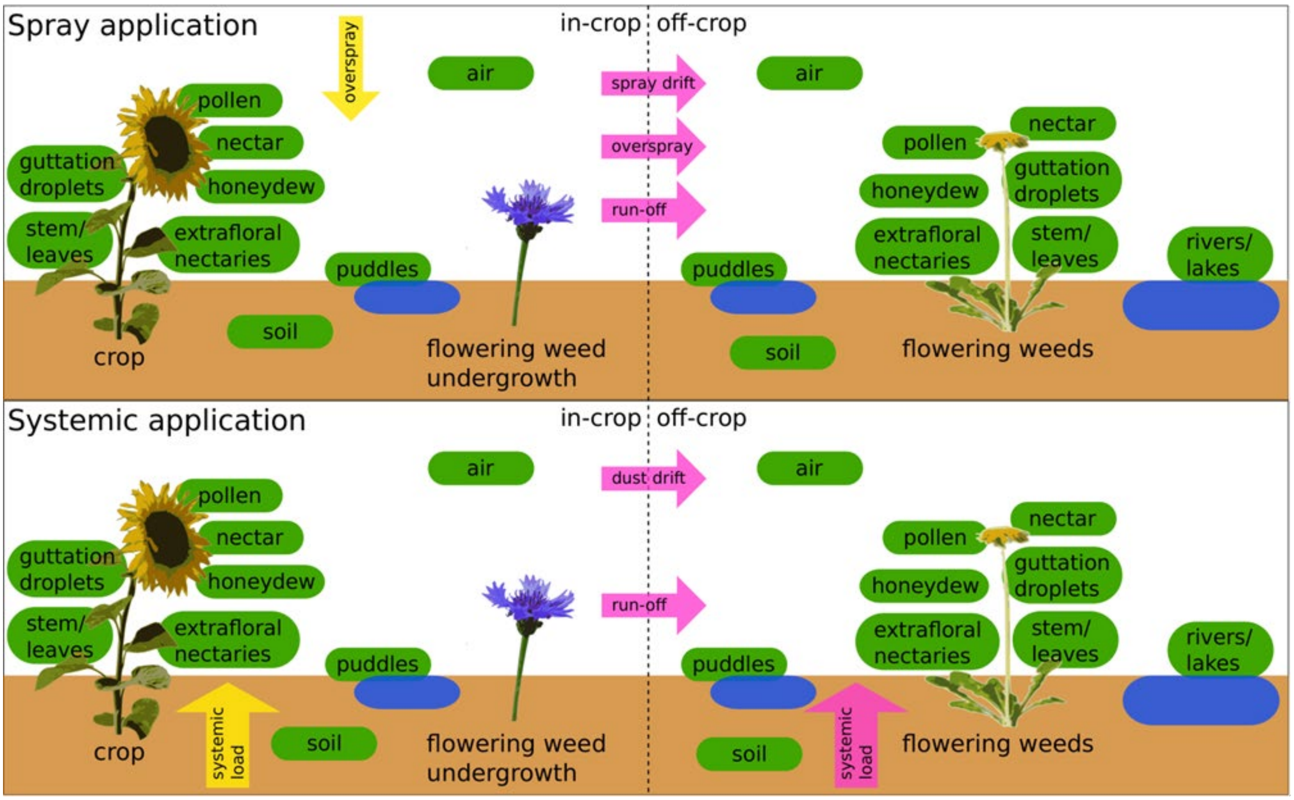
4.2.4 Soil

Agricultural soils are exposed to the portion of sprayed pesticides (in-field) that is not intercepted by crop coverage. Spray drift and run-off processes displace spray mist and pesticide residues on plants and soil into off-crop areas. In the case of solid systemic pesticide applications the in-field soil is directly applied. These pesticides can be transported along in-field as well as into off-field habitats due to dust drift and run-off. Depending on their persistence some pesticides (e.g. neonicotinoids) may stay in the soil over multiple years or even accumulate there (Goulson 2013a; Bonmatin et al. 2015).

4.2.5 Water bodies

FVIs such as bees collect water to maintain their osmotic balance. Honey bees also use water to control the colony temperature or to prepare liquid food for their larvae (Kühnholz & Seeley 1997). Furthermore, solitary bees use water to soften hard soil while excavating (Michener 2007). They collect it from water sources such as guttation water, in-crop and off-crop small ephemeral water bodies (puddles) or larger water bodies (rivers or lakes) (EFSA 2013). Puddle water has been shown to be a relevant source of pesticide exposure for FVIs (Samson-Robert et al. 2014). Puddles are exposed by direct overspray or dust drift in in-field habitats. Off-crop water bodies (puddles, rivers, lakes) are contaminated by pesticides through field edge overspray, spray drift, run-off after spray application, dust drift and run-off after systemic applications (Neumann et al. 2002; Schulz 2004). Furthermore, guttation water from in-crop or of-crop plants may be exposed via uptake of systemic pesticides.

Figure 9: Exposure of FVI species habitats and environmental matrices by spray and systemic pesticide applications.



Footnote: Primary exposure processes are depicted as yellow, upwards/downwards and secondary processes as pink, sideward arrows (except systemic load on off-crop habitats after run-off). Potentially contaminated matrices are illustrated as green bubbles.

Source: own illustration

4.2.6 General considerations

Within treated fields there can be undergrowth of flowering wild plants (i.e. weeds). These crops include permanent cultures such as vineyards and fruit orchards but also arable cereal crops. Undergrowth weeds are exposed by spray applications just as the crop. Therefore, there might be pesticide exposure of FVIs in crops that are not deemed attractive because of attractive interspersed weeds (e.g. cornflower or poppy species in cereal fields). Furthermore, due to the high soil persistence of some systemic pesticides (e.g. neonicotinoids) succeeding attractive crops (and weeds) might be exposed to accumulated residues from applications in previous cropping seasons (Goulson 2013b; Bonmatin et al. 2015). Moreover, pesticide applications in non-attractive crops can still lead to exposure of attractive off-crop habitats through secondary exposure processes (spray/dust drift, field edge overspray, run-off). In cases where systemic pesticides are applied as a spray they reach all habitat matrices through contact but also build up systemic loads in the soil which are taken up by crops and weeds (in-field) and off-field non-target plants. Most of the listed exposure pathways to specific habitat matrices have been discussed in the EFSA bee guidance document (EFSA 2013) except exposure of stem/leaves of crops and weeds/non-target plants. This matrix is relevant for all FVI species with herbivore life stages (e.g. moths, butterflies, beetles).

4.3 Conclusion

In-field habitats of FVI species can be exposed to pesticides through intended primary processes which include spray and solid systemic application but also stem application and irrigation whereas unintended secondary processes contaminate in-field and off-field habitats through spray drift, field edge overspray, dust dispersion/drift and run-off. Several in-field and off-field habitat compartments can take up pesticides by transport processes which include airspace, plant matrices (e.g. pollen, nectar, stem/leaves, guttation water, extrafloral nectaries, honey dew), soil and water bodies (e.g. puddles, rivers, lakes).

5 Pesticide exposure of flower-visiting insect species

Philipp Uhl, Carsten Brühl

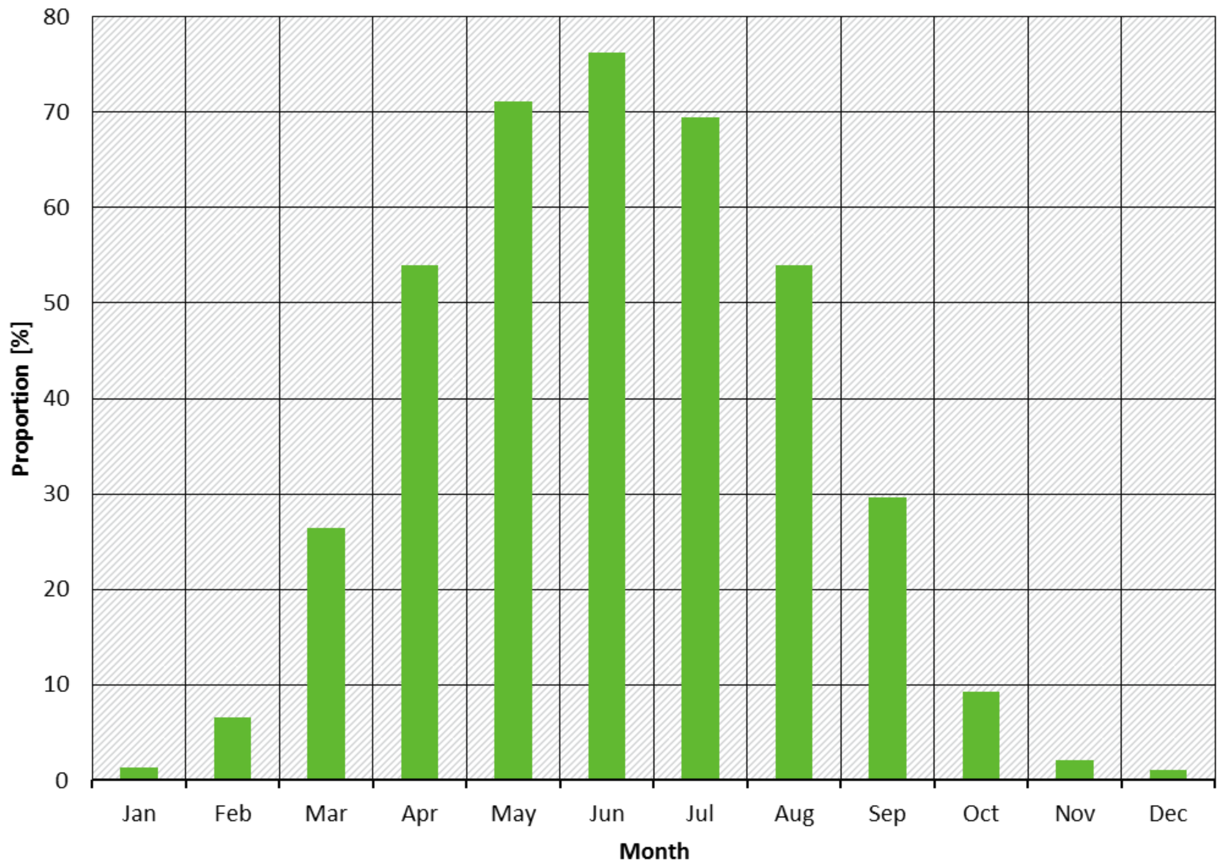
5.1 Exposure-relevant ecological traits

5.1.1 Bees

5.1.1.1 Flight period & duration

Throughout the year bee species flight activity varies considerably (Figure 10). In Europe, as early as February some bee species begin to actively forage. From March to September at least 25% of bee species are simultaneously active, from May to July around 70%. June is the month with the highest number of flying species (76%) (data from Roberts et al. unpublished). The active flight period of 96% of all European bee species ranges from one to seven months (Figure 11). A quarter of all species fly for three months, whereas three quarters are active between two and five months.

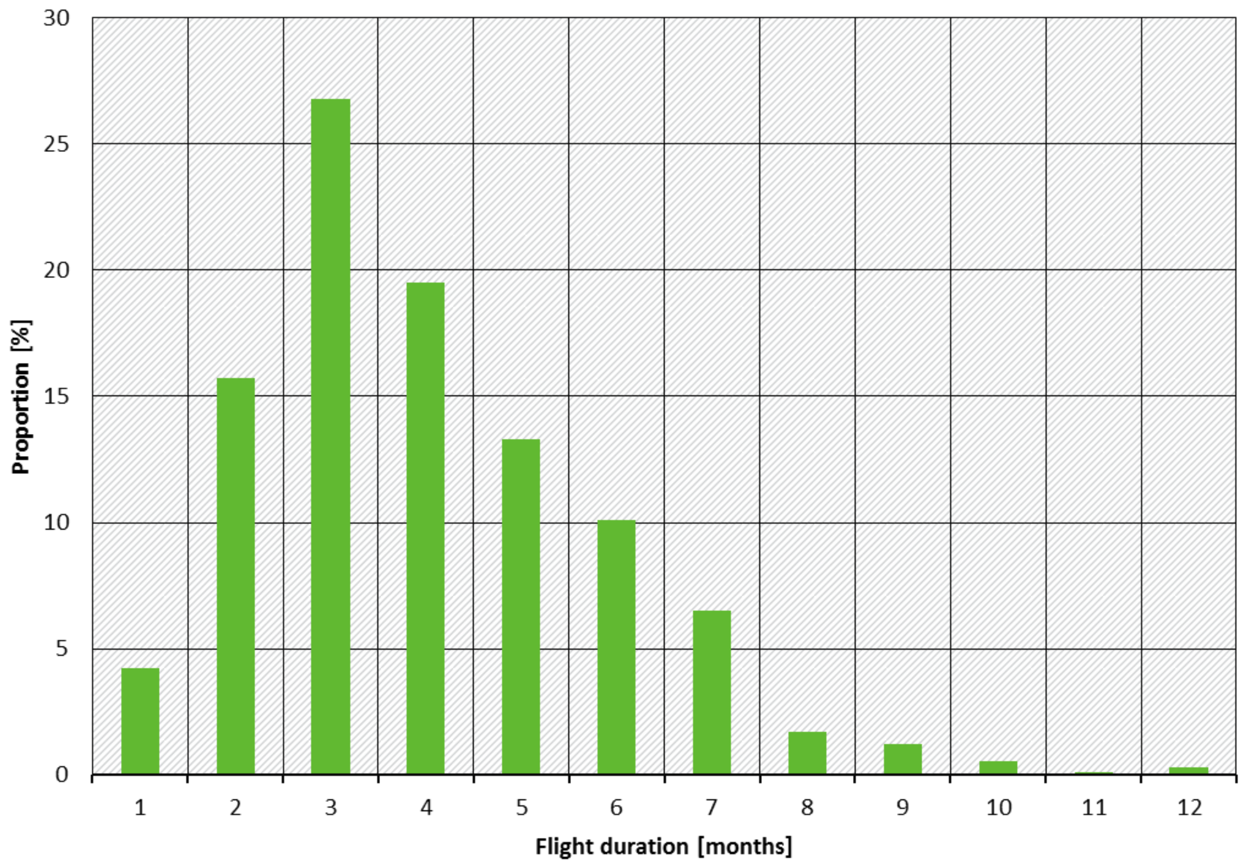
Figure 10: Flight activity of European bee species during the year.



Footnote: Dataset includes 1061 of 1936 species.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

Figure 11: Length of active flight period of European bee species.



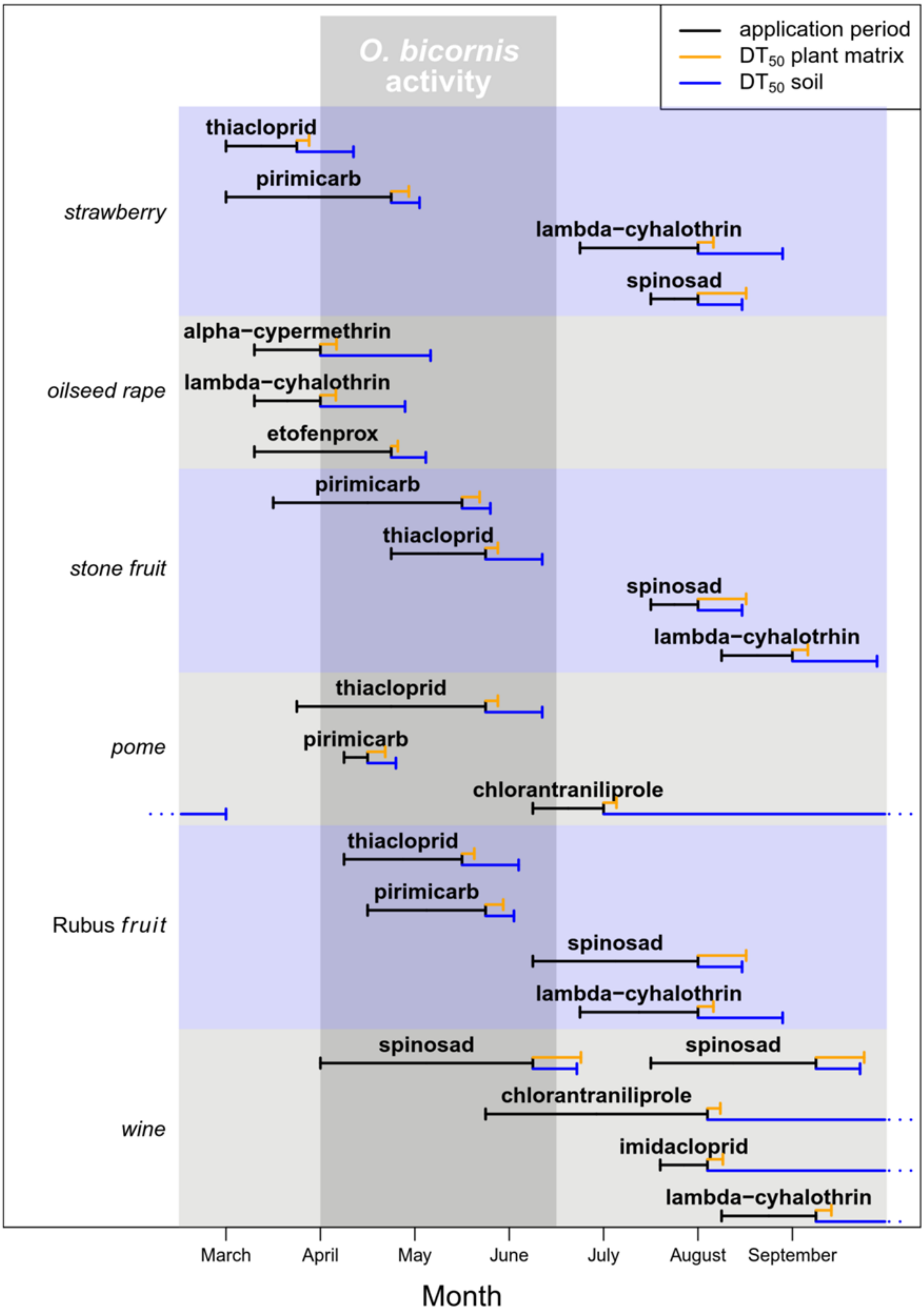
Footnote: Dataset includes 1061 of 1936 species.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

The annual flight period of wild bee species can coincide with pesticide applications in crops. Depending on the crop bee species might therefore be exposed to a wide variety of pesticides. A case study of Schulz (2016) showed for six German crops that the red mason bee *Osmia bicornis* is actively foraging in a time corridor where there is possible exposure to multiple insecticide products (Figure 12). *Osmia bicornis* adults might come in contact with these insecticides promptly after application via overspray/spray drift/dust drift or when foraging on exposed plants. Since it is a polylectic species all selected crops are potential forage. Additionally, there is potential exposure to pesticides that are applied outside its active flight period due to uptake of persistent compounds in soil and plant material while foraging. *Osmia bicornis* females for example collect mud to seal their brood cells. A majority of bee species also build their nest in soil (more than 64%; see below).

From March to September a considerable proportion of bee species (25%) simultaneously have their active flight period which lasts two to five month in most of these species (75%). In these months from spring to early autumn there is a high chance of exposure to pesticide applications since this is also the growing period of most agricultural crops in Central Europe. The core flight time of European wild bee species is from May to July where the vast majority of bees are actively foraging (76%). Additionally, many wild bee species can also be exposed to pesticides that were applied beforehand due to longer half lives in plants and soil of certain substance classes (e.g. neonicotinoids) (Fantke & Juraske 2013; Goulson 2013b; Bonmatin et al. 2015).

Figure 12: Application dates and persistence of several insecticides in selected German crops combined with the active flight period of *Osmia bicornis*.

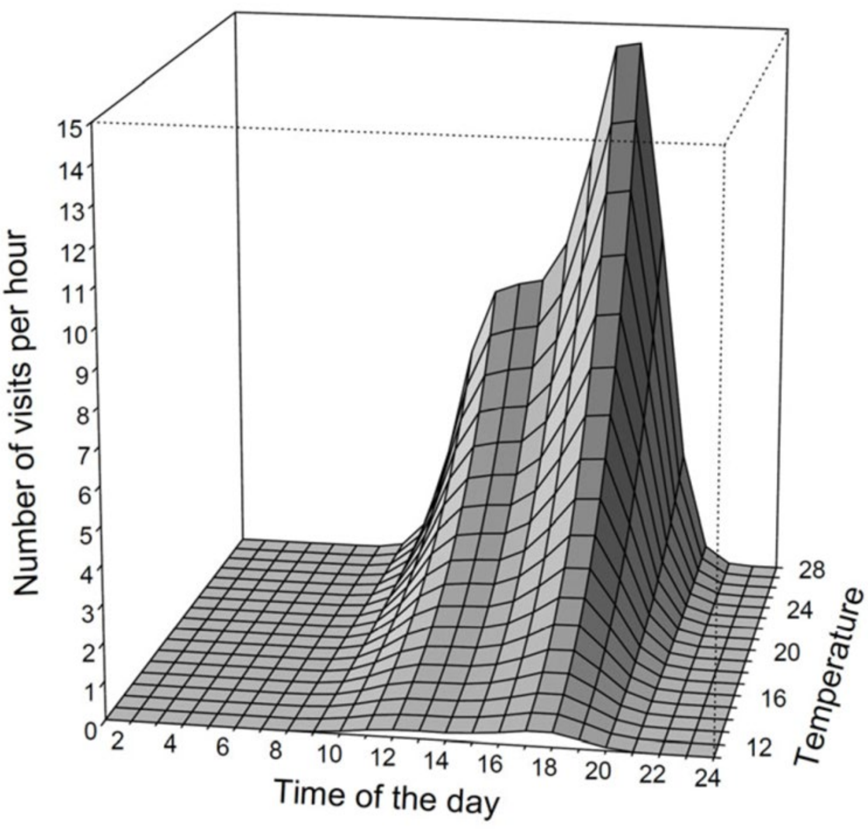


Footnote: Data are summarised from application recommendations from regional German authorities (south-west Germany) for pomiculture, viticulture and arable farming in 2016. Source: Schulz (2016)

Apart from their annual activity window bees also differ in their diel activity patterns. Thompson & Hunt (1999) reviewed the literature for bumble bees and found that activity is highest before 10:00 and after 16:00. In contrast, the number of foraging honey bee workers usually peaks midday. Bumble bees start foraging earlier than honey bees and finish later. They hypothesised that bumble bees can better cope with colder temperatures and also gain competitive advantage within these time intervals (Thompson & Hunt 1999). In honey bees there are two major parameters that affect activity: Forager departure from the colony is positively correlated with temperature and solar radiation. However, there seems to be a transition point with solar radiation when forager numbers decrease (Burrill & Dietz 1981). In a more recent study Steen (2016) recorded foraging activity of Norwegian bumble bee species on white clover *Trifolium repens* in July and August and modelled their diel activity (Figure 13). The data showed the start of foraging activity to be around 9:00 with a following rapid increase between 10:00 and 12:00. After a depression period activity increased again from 15:00 to 17:00, peaked between 17:00 and 18:00 and decreased strongly after 21:00. Furthermore, activity was correlated with temperature. In a flower visitor study on broadleaved lavender (*Lavandula latifolia*) it was shown that diel activity can vary greatly between bee species (Herrera 1990). Between and within bee families there is no clear general pattern emerging. Number and time of activity peaks seem species-specific and related to location and weather conditions (e. g. temperature, insolation).

Therefore, it is complicated to make qualified statements about potential exposure of wild bee species considering their daily flight activity. Delaying pesticide application into evening hours to minimise honey bee exposure might maximise bumble bee exposure. Since diel activity is also correlated with temperature, chances of exposure are generally higher on warmer days within a season.

Figure 13: Modelled diel activity of bumble bees foragers on white clover in summer.

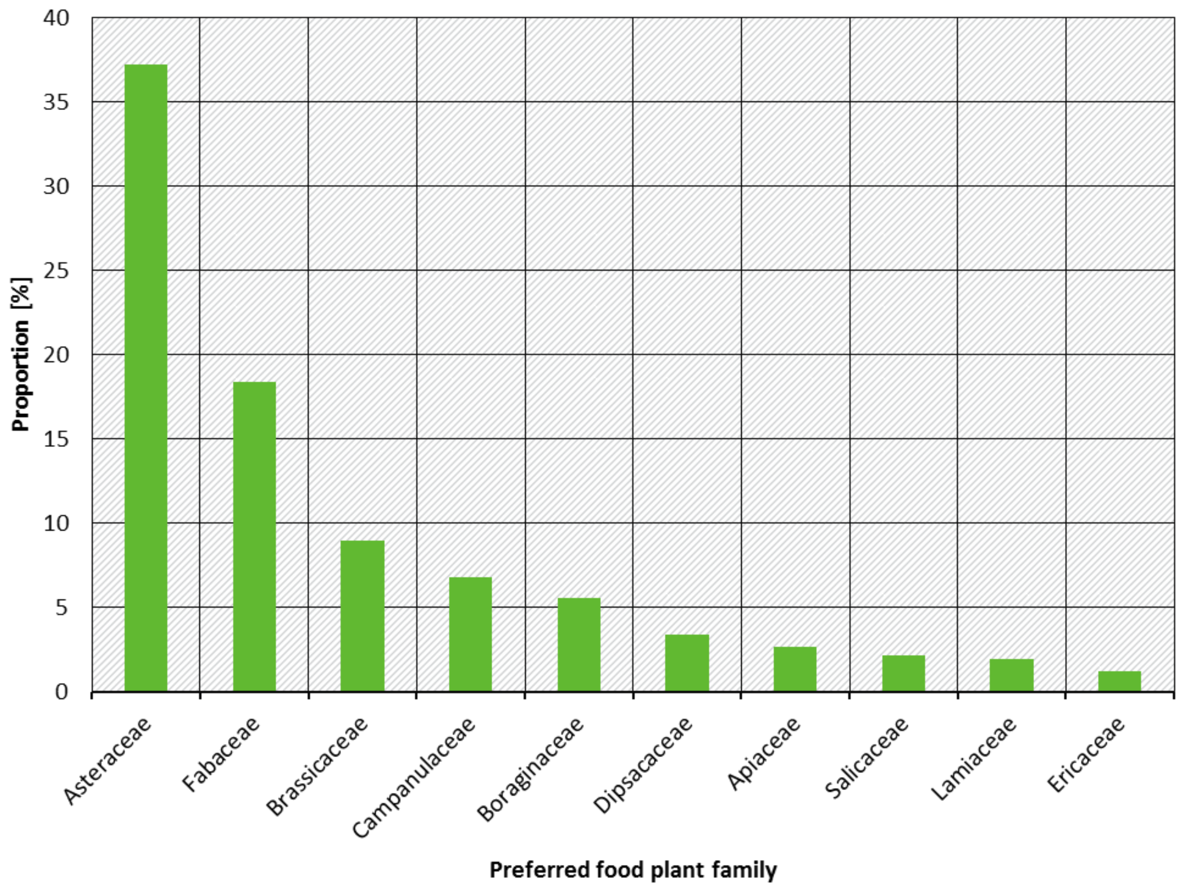


Source: Steen (2016)

5.1.1.2 Food plants

The preference of wild bee species to collect nectar and pollen from certain plant species could be assessed primarily for oligolectic species since they are specialised on single plant families or even individual plant species. Polylectic species do often also show preferences but are not dependent on specific food plants to successfully populate a habitat. Furthermore, it is harder to identify preferred food plants in polylectic species (Westrich 1990). Therefore, oligolectic species are far more susceptible to exposure from their preferred food plants that grow in in-field and off-field areas due to their narrow food niche. Oligolectic species make up 40% of all European bee species (Roberts et al. unpublished). Due to missing data concerning food plants only 21% of all bee species are included in this analysis (data for polylectic species with known food preferences (13% of subset) was included to improve the small dataset). Moreover, the taxonomic resolution varies greatly in the dataset. Therefore, plant data was scaled up to family level for analysis (Figure 14). Out of 31 plant families that are listed in the European bee trait database (Roberts et al. unpublished) the family preferred by most bee species was Asteraceae (37%) followed by Fabaceae (18%). The presence of wild bee species in flowering leguminous crops (Fabaceae such as broad bean *Vicia faba*, common bean *Phaseolus vulgaris* or alfalfa *Medicago sativa*; see Table 2; EFSA list of attractive crops) as well as sunflowers (Asteraceae) is therefore highly likely and needs special attention in risk assessment.

Figure 14: Preferred food plant families of European bee species (10 most common families).



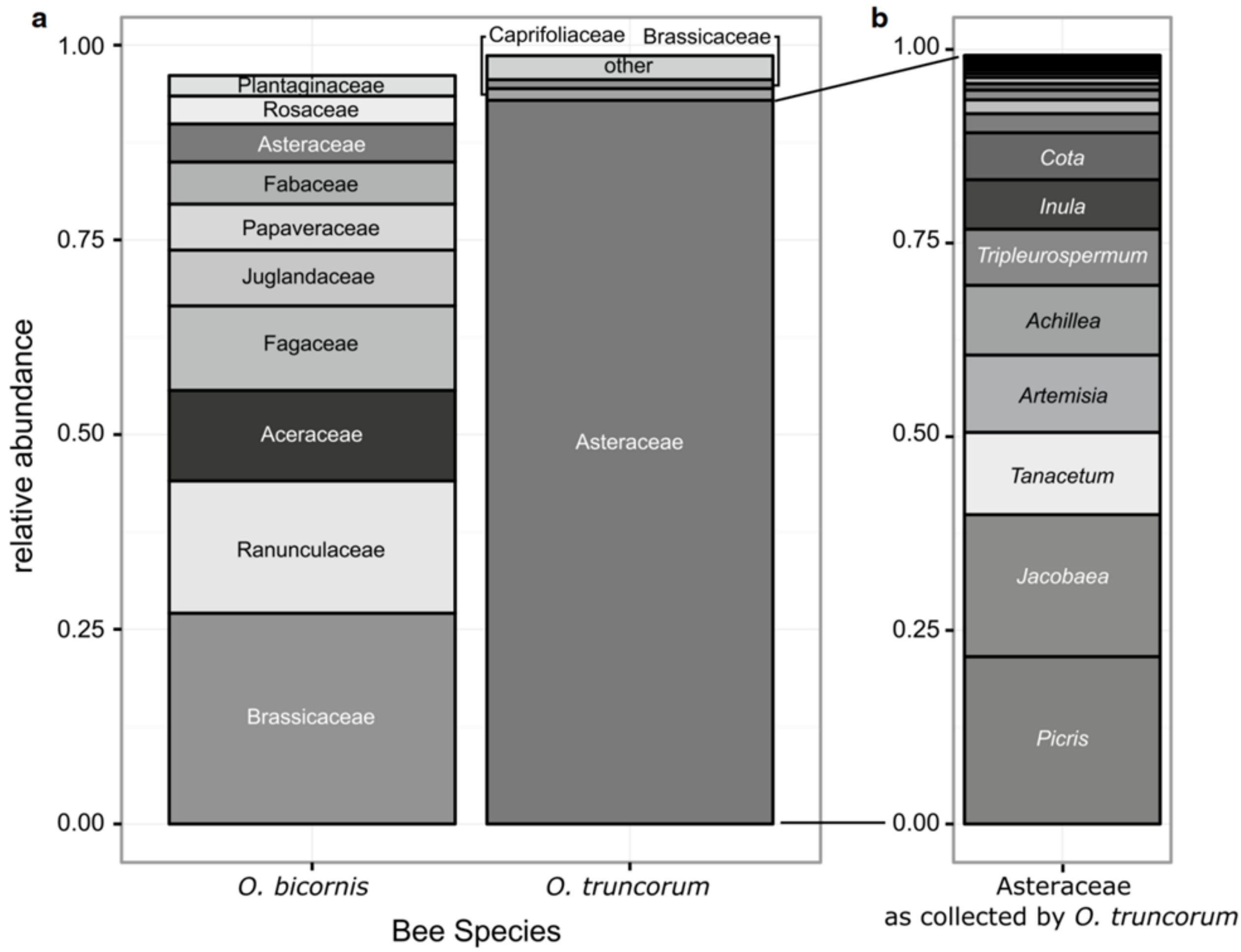
Footnote: Data set includes 414 of 1936 species.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

It is difficult to pinpoint a favoured food source for all generalist and semi-specialist bee species. It was found that the polylectic *O. bicornis* collects pollen from plants of multiple families (Figure 15). The more specialised *Osmia truncorum* collects mostly Asteraceae pollen but from several plant genera (Sickel et al. 2015). Moreover, bees do not necessarily collect nectar and pollen from the same plant species. In a field study honey bees collected nectar mainly from oilseed rape and sunflower but pollen from a multitude of weeds, herbaceous and woody plants (Requier et al. 2015). There is still only data concerning the diet spectrum of a few bee species. Generally, polylectic bee species may be exposed to pesticide residues from a wider flower spectrum than oligolectic species. Therefore, these species are more likely to be exposed to higher residue doses in mass-flowering crops.

For German wild bee species oilseed rape *Brassica napus* (11% of species), blackberry *Rubus fruticosus* (8%) and field mustard *Sinapis arvensis* (6%) are listed as the crops that are visited by most species (Westrich 1990). Moreover, there are several crop-associated wild plants that grow in off-field areas and are attractive to many bee species (data analysed from Westrich (1990)): Common dandelion *Taraxacum officinale* (24% of German bee species), clover *Trifolium ssp.* (16%), thistle *Cirsium ssp.* (15%), knapweed *Centaurea ssp.* (15%), buttercup *Ranunculus ssp.* (9%), common yarrow *Achillea millefolium* (9%) and wild carrot *Daucus carota* subsp. *carota* (9%). It has to be noted that these data from Westrich (1990) are based on observations not on quantitative measurements of collected pollen.

Figure 15: Pollen spectrum of *O. bicornis* and *O. truncorum*.



Footnote: a) Ten most abundant families as collected by both bee species. For *O. truncorum* “other” includes the families Apiaceae, Rosaceae, Fabaceae, Ranunculaceae, Plantaginaceae, Juglandaceae and Amaranthaceae. b) Plant genera detected within the Asteraceae collected by *O. truncorum*. For visualisation reasons, only the eight most abundant genera are labelled.

Source: Sickel et al. (2015)

5.1.1.3 Nesting

In contrast to food plants data on the preference of specific nesting habits of European bees are quite extensive (information for 83% of all species in the European trait database; Roberts et al. unpublished). The majority of bee species (64%) dig into the ground to build their nests ("ground excavator"). Parasites are the second biggest group (23%). They lay their eggs into brood cells of solitary bees or live within the colonies of social bee species (Westrich 1990). Therefore, they assume the nesting trait of their host. Considering that most bee species are ground excavators it is very likely that most parasitic species have ground excavators as hosts. In German bee species 74% of all hosts are ground excavators (data analysed from Westrich 1990). The remaining bee species (13%) are mainly "renters" which mean that they lay their eggs in existing empty cavities (e.g. beetle holes in deadwood or snail shells) but also "aboveground excavators", "masons" (use of mud to fashion entire cell) and "carders" (use shredded plant material) (definitions are adopted from the European bee trait database; Roberts et al. unpublished).

Since ground excavators are the biggest group soil exposure should be recognised as a critical pesticide exposure pathway. Furthermore, bee species from other groups can also be exposed through soil contact: e.g. *O. bicornis* as a renter (by the definition of Roberts et al. unpublished) seals its brood cells with collected mud. Furthermore, some bee species can also be exposed by plant material that they collect for nest building: e.g. leaf cutter bees (*Megachile* ssp.) gather leaf material from different in-crop weeds and off-crop non-target plants (Westrich 1990).

5.1.1.4 Sociality

Brittain & Potts (2011) hypothesised that social bee species might have a higher probability to be exposed to pesticides than solitary bees. Due to the sheer number of foragers they can bring back pollen and nectar of many different plant species and from a wider area to their colonies. This may lead to higher residue levels in stored food and larval provisions and to exposure to multiple pesticides. However, further evidence is needed to support this claim. Residue data of stored food items is mostly available only for the honey bee (see section 5.2 Evidence of exposure by residue levels of pesticides).

5.1.1.5 Mobility

It is conceivable that highly mobile species experience lower pesticide exposure due to dilution effects. Since their foraging grounds are relatively vast they might take up less pesticides from a specific field or orchard as less mobile species. However, it is also likely that they are exposed to a wider variety of pesticides (mixtures) and therefore higher overall doses than species with low mobility which do not forage in different crops (de Palma et al. 2015). Therefore, a dilution effect cannot be assumed as a realistic worst case scenario. However, there is a need for further research to address these hypotheses. Such investigations should take into account that the distribution of bee foraging ranges is skewed left: 70% of all European bee species have foraging ranges of 1 km and lower, 50% of ca. 500 m and lower (data analysed from European bee trait database; see Definition of ecologically vulnerable groups, p. 79; Roberts et al. unpublished).

5.1.2 Moths & butterflies

The assessment of exposure-relevant traits in Lepidoptera is more difficult than in bees since there is no comprehensive database of butterfly or moth traits available (see 3.2 Categories derived from ecological traits). Therefore, ecological attributes of lepidopteran species can be listed but their relevance for exposure incidence in butterfly and moth communities need to be evaluated in follow-up studies.

A major ecological difference in lepidopterans is their activity time during the day. Diurnal species have a substantially higher chance to be directly exposed to pesticides during spray applications whereas nocturnal species (most moths species) often hide at daytime minimising their direct exposure (Longley & Sotherton 1997). Adult preference for crops as food plants may increase chances of in-crop exposure. However, the majority of species feed on non-crop plants (Ebert 1994; Scoble 1995). Furthermore, host plant specialisation of lepidopteran caterpillars could have an impact on their exposure, especially in cases where caterpillars are actually crop pests (e.g. small white *Pieris rapae* feeding of cabbage) that are targeted with pesticides. Moreover, habitat specialists that live in grassy field edge habitats are more likely to be exposed than species that prefer forest edges. As in bees their flight phase may coincide with pesticide applications in certain crops and longer flight durations should increase the likelihood of exposure. Adult mobility can also play a role but as it is the case with bees it remains unclear if sedentary or migratory strategies might increase exposure.

5.2 Evidence of exposure by residue levels of pesticides

5.2.1 Individuals

Investigations of pesticide residues levels in FVI individuals are indispensable to assess pesticide exposure. Unfortunately, these data are only available for bees at the moment. Data for other FVI groups such as lepidopterans, flies and beetles should be collected to allow for adequate exposure assessment. Bees are exposed to a plethora of pesticides. Mullin et al. (2010) analysed 140 brood and adult bee samples from North American honey bee colonies and detected 46 pesticides of different pesticide classes and their metabolites (Table 20). They found up to 25 different pesticides in a single sample (2.5 on average). 30 of these substances have insecticidal properties or are metabolites of insecticides. The two most frequently detected pesticides were in-hive acaricides fluvalinate and coumaphos. Chauzat et al. (2011) sampled honey bee colonies in France between 2002 and 2005 and detected residues of 25 substances. A comprehensive list of pesticides that were detected in individual adult honey bees was compiled by EFSA PPR Panel (2012). This list includes all major pesticide classes (e.g. insecticides, fungicides, herbicides).

The predominant part of bee exposure studies in recent years is investigating the chemical 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 & Goulson 2017). Therefore, in the following sections such research is overrepresented compared to other pesticide classes or bee species.

Table 20: Pesticide and metabolite detections in individual honey bees from North American honey bee colonies.

Pesticide	Pesticide class	Activity	Mean dose	Dose range	LOQ	LOD
			(ng/g)			
Fluvalinate	Pyrethroid	I, A	357.7	1.1-5860.0	NA	1.0
Coumaphos	Organophosphate	I, A	50.4	1.0-762.0	NA	1.0
Chlorpyrifos	Organophosphate	I, A	3.4	1.0-10.7	NA	0.1
Chlorothalonil	Aromatic fungicide	F	100.2	1.5-878.0	NA	1.0
Cypermethrin	Pyrethroid	I, A	10.1	2.0-25.8	NA	1.0
Permethrin	Pyrethroid	I, A	2478.1	12.0-19600.0	NA	10.0
DMPF (amitraz)	Formamidine	I, A	1249.1	6.0-9040.0	NA	4.0
Esfenvalerate	Pyrethroid	I	4.3	1.0-9.3	NA	0.5
Methidathion	Organophosphate	I	16.2	6.5-32.0	NA	1.0
Deltamethrin	Pyrethroid	I	29.3	23.0-39.0	NA	20.0
Pendimethalin	Dinitroaniline	H	15.9	6.5-27.6	NA	1.0
Cyfluthrin	Pyrethroid	I	8.2	2.0-14.0	NA	1.0
Dicofol	Organochlorine	A	2.1	1.0-3.8	NA	0.4
Fenpropathrin	Pyrethroid	I, A	17.1	2.8-37.0	NA	0.4
Azinphos methyl	Organophosphate	I, A	13.3	4.8-22.0	NA	3.0
Cyprodinil	Anilinopyrimidine	F	12.6	9.2-19.0	NA	5.0
THPI (captan)	Phthalimide	F	40.2	37.7-43.4	NA	30.0
Allethrin	Pyrethroid	I	16.6	6.7-24.0	NA	1.0
Tetramethrin	Pyrethroid	I	21.3	18.0-23.0	NA	6.0
Methoxyfenozide	Insect growth regulator	I	8.6	1.5-21.0	NA	0.4

Pesticide	Pesticide class	Activity	Mean dose	Dose range	LOQ	LOD
				(ng/g)		
Endosulfan I	Cyclodiene	I	3.0	1.3-6.1	NA	0.1
Endosulfan sulfate	Cyclodiene	I	2.4	1.6-3.0	NA	0.1
Endosulfan II	Cyclodiene	I	1.9	1.4-2.4	NA	0.1
Parathion methyl	Organophosphate	I, A	1.8	1.5-2.0	NA	1.0
Cyhalothrin	Pyrethroid	I, A	1.5	1.1-1.8	NA	0.1
DMA (amitraz)	Formamidine	I, A	2507.5	275.0-4740.0	NA	50.0
Fipronil	Phenylpyrazole	I, A	1535.0	9.9-3060.0	NA	1.0
Bifenthrin	Pyrethroid	I, A	7.6	2.9-12.3	NA	0.4
Dieldrin	Cyclodiene	I	11.0	10.0-12.0	NA	4.0
Prallethrin	Pyrethroid	I	7.4	6.2-8.6	NA	4.0
Coumaphos oxon	Organophosphate	I, A	4.5	2.1-6.8	NA	5.0
Oxyfluorfen	Nitrophenyl ether	H	4.3	3.8-4.8	NA	0.5
Chlorfenapyr	Pyrrole	I, A	2.3	1.8-2.7	NA	1.0
Carbaryl	Carbamate	I, F	588.0	588.0-588.0	NA	5.0
1-Naphthol (carbaryl)	Carbamate	I	238.0	238.0-238.0	NA	2.0
Dimethomorph	Morpholine	F	56.0	56.0-56.0	NA	15.0
Tebuconazole	Conazole	F	34.0	34.0-34.0	NA	3.0
Chlorferone (coumaphos)	Organophosphate	I, A	25.0	25.0-25.0	NA	25.0
Tebufenozide	Insect growth regulator	I	23.0	23.0-23.0	NA	2.0
Fenoxaprop-ethyl	Aryloxyphenoxy-propionic	H	15.4	15.4-15.4	NA	6.0
Atrazine	Chlorotriazine	H	15.0	15.0-15.0	NA	1.0
Carbendazim	Benzimidazole	F	14.3	14.3-14.3	NA	1.0
Pyraclostrobin	Methoxycarbani-late strobilurin	F	8.6	8.6-8.6	NA	1.0
DDE p,p'	Organochlorine	I	6.6	6.6-6.6	NA	3.0
Fluridone	NA	H	6.5	6.5-6.5	NA	5.0
Pronamide	Amide herbicide	H	2.2	2.2-2.2	NA	1.0

Table adapted from Mullin et al. (2010). Sorted by number of detections. Column "activity": A – acaricide, F – fungicide, H – herbicide, I – insecticide. LOD and LOQ values were rounded to one digit. Doses reported as <LOD or >LOD were standardised as equal LOD, doses reported as <LOQ as equal LOQ and doses reported as <"value" as equal to this value.

5.2.2 Nectar & Pollen

5.2.2.1 Crops

Nectar and pollen are major carriers of pesticide loads for FVIs. In their residue study of honey bee colonies Mullin et al. (2010) found 98 pesticides and degradates in collected pollen (350 samples). In a three year long study of French honey bee colonies 19 pesticides were detected in pollen collected with pollen traps (Chauzat et al. 2006). The by far most frequently found substances were imidacloprid (50%) and its metabolite 6-chloronicotinic acid (44.4%) (Table 21). For an exhaustive list of residues of different pesticides in pollen please refer to EFSA PPR Panel (2012).

Table 21: Pesticide residues in pollen samples of French honey bee colonies.

Pesticide	Pesticide class	Activity	Mean dose	Dose range	LOQ	LOD
				(ng/g)		
Carbofuran	Carbamate	I, A	14.0	5.0-10.9	10.0	5.0
Oxamyl	Carbamate	I, A	38.4	38.4-38.4	10.0	5.0
Carbaryl	Carbamate	I, F	218.7	126.0-265.0	10.0	5.0
Imidacloprid	Neonicotinoid	I	1.2	0.2-5.7	1.0	0.2
6-Chloronicotinic acid	Metabolite of imidacloprid	I	1.2	0.2-9.3	0.6	0.2
Endosulfan	Organochlorine, cyclodiene	I	81.2	0.1-340.0	8.0	0.1
Parathion-ethyl	Organophosphate	I	19.2	8.0-30.4	30.4	8.0
Parathion-methyl	Organophosphate	I	24.8	10.0-39.5	39.5	10.0
Coumaphos	Organophosphate	I, A	925.0	150.0-1700.0	142.6	37.0
Fipronil	Phenylpyrazole	I	1.2	0.3-(2.0-0.5)	2.0-0.5	0.3
Fipronil sulfone compound	Metabolite of fipronil	I	1.6	0.3-3.7	2.0-0.5	0.3
Fipronil desulfynil compound	Metabolite of fipronil	I	1.3	0.3-1.5	2.0-0.5	0.3
Tau-fluvalinate	Pyrethroid	I, A	487.2	1.1-2020.0	76.0	1.1
Cyproconazole	Conazole	F	7.5	5.0-10.0	10.0	5.0
Tebuconazole	Conazole	F	12.3	12.3-12.3	10.0	5.0
Myclobutanil	Conazole	F	13.9	5.0-20.3	10.0	5.0
Hexaconazole	Conazole	F	18.0	18.0-18.0	20.0	10.0
Flusilazole	Conazole	F	26.1	5.0-71.0	10.0	5.0
Penconazole	Conazole	F	27.6	5.0-126	10.0	5.0

Table adapted from Chauzat et al. (2006). Column "activity": A – acaricide, F – fungicide, I – insecticide. Doses reported as <LOD or >LOD were standardised as equal LOD, doses reported as <LOQ as equal LOQ and doses reported as <"value" as equal to this value.

In a review of the environmental impact of neonicotinoids Wood & Goulson (2017) compiled expected residues of three neonicotinoids in pollen and nectar of selected crops (calculated by EFSA, Table 22). Pollen doses are consistently higher than nectar doses. Furthermore, residue levels fluctuate between crops. Similar maximum residue levels of neonicotinoids (nectar: 1.9 ng/g, pollen: 6.1 ng/g) were calculated by Godfray et al. (2014) from a revision of 20 studies. Goulson (2013a) also performed a literature review and quantified the range of neonicotinoid in nectar as <1.0 to 8.6 ng/g and in pollen as <1.0 to 51.0 ng/g.

Table 22: Summary of expected residues in pollen and nectar of various neonicotinoid-treated flowering crops.

Crop	Pesticide	Residues in pollen (ng/g)		Residues in nectar (ng/g)	
		Minimum	Maximum	Minimum	Maximum
Oilseed rape	Clothianidin	5.95	19.04	5.00	16.00
Sunflower	Clothianidin	NA	3.29	NA	0.32
Maize	Clothianidin	7.38	36.88	NA	NA
Oilseed rape	Imidacloprid	1.56	8.19	1.59	8.35
Sunflower	Imidacloprid	NA	3.90	NA	1.90
Maize	Imidacloprid	3.02	15.01	NA	NA
Cotton	Imidacloprid	3.45	4.60	3.45	4.60
Oilseed rape	Thiamethoxam	4.59	19.29	0.65	2.72
Sunflower	Thiamethoxam	2.38	3.02	0.59	0.75
Maize	Thiamethoxam	13.42	21.51	NA	NA

Table adapted from Wood & Goulson (2017). Calculated by EFSA from the review of outdoor field trials. No nectar values are available for maize as this plant does not produce nectar.

Blacquiere et al. (2012) reported a range of average imidacloprid pollen residues values of 0.9-3.1 ng/g (summary of eight studies). In a German study, in which pollen was sampled from apiaries, thiacloprid was detected in 33% of all samples (max values of 199 ng/g)(Genersch et al. 2010). Since 2014 several neonicotinoid pollen and nectar residue studies have been published. Those which analysed bee-collected pollen and nectar were summarised by Wood & Goulson (2017) (Table 23): In many of these studies oilseed rape pollen and nectar were analysed. When directly sampling oilseed rape flowers, Botías et al. (2015) found mean pollen doses of 3.26 ng/g of thiamethoxam, 2.27 ng/g of clothianidin and 1.68 ng/g of thiacloprid. Nectar samples contained mean doses of 3.20 ng/g of thiamethoxam, 2.18 ng/g of clothianidin and 0.26 ng/g of thiacloprid. Xu et al. (2016) detected clothianidin at 0.6 ng/g in nectar of oilseed rape in the Midwestern USA and Western Canada. In honey bee-collected nectar neonicotinoid doses of 0.68-0.77 ng/mL were found when honey bee colonies were set up adjacent to neonicotinoid seed-treated oilseed rape fields. Colonies in the vicinity of non-seed-treatment plants showed lower doses of <0.3 ng/mL (Rolke et al. 2016). In seed-treated maize pollen thiamethoxam and clothianidin were found at mean levels of 1 ng/g (LOD) to 5.9 ng/g (Stewart et al. 2014). Xu et al. (2016) reported a similar clothianidin dose of 1.8 ng/g in their study.

There have been very few residue studies with other species than the honey bee. Cutler & Scott-Dupree (2014) detected low clothianidin doses at average 0.4 ng/g and thiamethoxam at <0.1 ng/g (LOD) in maize pollen collected by *Bombus impatiens*. In a study with *Bombus terrestris* mean clothianidin levels in oilseed rape pollen were measured at 0.88 ng/g (Rolke et al. 2016). In contrast, Rundlöf et al. (2015) found much higher levels of 5.4 ng/mL in bumble bee-collected oilseed rape nectar and nectar collected by honey bees (10.3 ng/mL). Only one study analysed clothianidin residues in pollen collected by a solitary bee (*Osmia bicornis* foraging on oilseed rape) which were on average 0.88 ng/g compared to residues of 0.50 and 0.97 in pollen collected from honey bees (Rolke et al. 2016).

Levels of neonicotinoid residues of bee-collected pollen and nectar vary by one order of magnitude between studies. Wood & Goulson (2017) concluded that several parameters such as dose and mode of treatment, studied crop, season, location, soil type, weather and time of day samples were collected influence neonicotinoid doses in both matrices. They further stated that these levels from recent field studies are in line with EFSA findings (Table 22) and results of Godfray et al. (2014). There are, however, still only a few data on wild bee-collected residues in pollen and nectar. Pesticide residue levels in crop pollen and nectar are also applicable to the exposure assessment of other FVI groups (e.g. lepidopterans, flies and beetles). For further information see the following review articles on pesticide residues in floral matrices: Blacquière et al. (2012), Bonmatin et al. (2015), EFSA (2012), Godfray et al. (2014, 2015), Goulson (2013a), Sanchez-Bayo & Goka (2014) and Wood & Goulson (2017).

Table 23: Neonicotinoid residues in pollen and nectar collected by different bee species in field studies.

Bee species	Matrix	Adjacent agriculture	Pesticide	Mean dose	Dose range	LOQ	LOD	Comment	Reference
				(ng/mL or ng/g)					
Apis mellifera	Nectar	Oilseed rape	Thiamethoxam	0.7-2.4	NA	0.5	NA	Range of reported median doses, bee stomach or hive sample	Pilling et al. (2013)
Apis mellifera	Nectar	Oilseed rape	Clothianidin	1.31	0.3-3.6	1.0	0.3	Tunnel study, bee stomach	Rolke et al. (2016)
Apis mellifera	Nectar	Oilseed rape	Clothianidin	0.68	0.3-1.6	1.0	0.3	Bee stomach	Rolke et al. (2016)
Apis mellifera	Nectar	Oilseed rape	Clothianidin	0.77	1.0-1.5	1.0	0.3	Bee stomach	Rolke et al. (2016)
Apis mellifera	Nectar	Oilseed rape	Clothianidin	10.30	6.7-16.0	0.5	0.2	Bee stomach	Rundlöf et al. (2015)
Apis mellifera	Pollen	Maize	Clothianidin	0.66	0.6-9.4	NA	0.1	Pollen trap samples	Long & Krupke (2016)
Apis mellifera	Pollen	Maize	Thiamethoxam	1.0-7.0	NA	1.0	NA	Range of reported median doses, bee bread or pollen load	Pilling et al. (2013)
Apis mellifera	Pollen	Oilseed rape	Clothianidin	13.90	6.6-23.0	1.5	0.5	Pollen load	Rundlöf et al. (2015)
Apis mellifera	Pollen	Oilseed rape	Clothianidin	0.84	NA	0.5	0.4	Pollen trap samples	Cutler et al. (2014)
Apis mellifera	Pollen	Oilseed rape	Thiamethoxam	1.0-3.5	NA	1.0	NA	Range of reported median doses, bee bread or pollen load	Pilling et al. (2013)
Apis mellifera	Pollen	Oilseed rape	Clothianidin	1.67	1.0-3.5	1.0	0.3	Tunnel study, pollen trap samples	Rolke et al. (2016)
Apis mellifera	Pollen	Oilseed rape	Clothianidin	0.50	0.3-1.1	1.0	0.3	Pollen trap samples	Rolke et al. (2016)
Apis mellifera	Pollen	Oilseed rape	Clothianidin	0.97	0.3-2.7	1.0	0.3	Pollen trap samples	Rolke et al. (2016)
Bombus impatiens	Pollen	Maize	Clothianidin	0.40	NA	0.5	0.1	Bee pollen load	Cutler & Scott-Dupree (2014)

Bee species	Matrix	Adjacent agriculture	Pesticide	Mean dose	Dose range	LOQ	LOD	Comment	Reference
				(ng/mL or ng/g)					
Bombus impatiens	Pollen	Maize	Thiamethoxam	0.10	NA	0.5	0.1	Bee pollen load	Cutler & Scott-Dupree (2014)
Bombus terrestris	Nectar	Oilseed rape	Clothianidin	5.40	1.4-14.0	0.5	0.2	Bee stomach	Rundlöf et al. (2015)
Bombus terrestris	Pollen	Oilseed rape	Clothianidin	0.88	0.3-1.3	1.0	0.3	Pollen load	Rolke et al. (2016)
Osmia bicornis	Pollen	Oilseed rape	Clothianidin	0.88	0.3-1.7	1.0	0.3	Pollen provision	Rolke et al. (2016)

Table adapted from Wood & Goulson (2017). Studies were chosen that simultaneously investigated residues in neonicotinoid seed-treated and non-seed-treated crops. Only residues of sites adjacent to neonicotinoid seed-treated crops are shown here for brevity reasons. Doses reported as <LOD or >LOD were standardised as equal LOD, doses reported as <LOQ as equal LOQ and doses reported as <"value" as equal to this value.

5.2.2.2 Wild plants

Bees collect pollen and nectar from a wide variety of plants. The question arose if foraging on wild plant flowers would dilute pesticide exposure or if wild plants are themselves major sources of pesticide residues. Since 2013 several studies have investigated pesticide levels in non-cultivated plants in agricultural surroundings (mostly neonicotinoid studies, Table 24 & Table 25). These studies show that vegetation in non-target areas (field margins adjacent to fields) is often contaminated with pesticides, too.

Botías et al. (2015) found neonicotinoid residues in several wild plant species next to seed-treated oilseed rape and wheat fields (Table 24): They detected thiamethoxam, imidacloprid and thiacloprid at greatly variable doses as high as 64 ng/g (*Papaver rhoeas*) and 86 ng/g (*Heracleum sphondylium*). Levels of neonicotinoids were much lower in nectar (mostly below LOD). In sown pollinator strips next to neonicotinoid-treated maize clothianidin residues between 0.2 and 1.5 ng/g were measured, with substantial differences between plant species (Mogren & Lundgren 2016). When summarised, all studies of wild plants since 2013 report mean levels of neonicotinoids in pollen from <0.4 to 14.8 ng/g and nectar from <0.1 to 1.5 ng/g (Table 24). These residue levels are comparable to residue levels found in seed-treated crops when flowering. However, since the database for wild plants is still small, such comparisons should be made with care. It would be also helpful to collate wild plant flower residue data of compounds that are directly applied in crops.

To link exposure of wild plant pollen to bee exposure it is necessary to analyse bee-collected pollen sources. Krupke et al. (2012) recorded clothianidin in pollen collected by honey bees. They could show that 55.5% of contaminated pollen loads were actually from wildflowers. In several other studies with free-flying honey bees as pollen collectors there are noticeable trends: The highest levels of residues are found when a large proportion of crop pollen is collected (Table 25). Pohorecka et al. (2013) and Rundlöf et al. (2015) found mean clothianidin doses higher than 10 ng/g in pollen loads with wildflower proportions of 73.7% and 37.9%, respectively. Conversely, when wildflower proportions in pollen are higher neonicotinoid residue doses are lower (Table 25). Botías et al. (2015) analysed honey bee pollen residue loads during and after oilseed rape bloom (91.1% and 100% wildflower, respectively) and found that total neonicotinoid content dropped from 3.09 to 0.20 ng/g.

There are only two studies which incorporated another bee species: Cutler & Scott-Dupree (2014) detected a very low proportion of maize pollen (<1%) collected by *Bombus impatiens* whose nests were placed next to a seed-treated maize field. This resulted in clothianidin mean levels of 0.40 ng/g in collected pollen. In a study by David et al. (2016) *B. terrestris* colonies were set up at farmland sites with an average distance of 590 m to treated oilseed rape. Pollen samples contained relatively high levels of thiamethoxam (18 ng/g) which might be attributable to a high proportion of oilseed rape pollen (31.9%). However, it is difficult to interpret the results for wild bees because of the low number of studies.

In general, all these studies show that high acute doses in pollen and nectar coincide with the bloom of FVI-attractive crops. However, chronic exposure of species with a long active flight period such as honey bees might be driven by wildflower foraging (Wood & Goulson 2017). Botías et al. (2015) calculated that 97% of total neonicotinoid residues in pollen in June and August were actually derived from wildflowers. In spite of the prevalence of studies pesticide residues in wildflowers are not restricted to systemic neonicotinoids. Long & Krupke (2016) detected 28 pesticides of different major insecticide classes (e.g. carbamates, organophosphates, pyrethroids) and also herbicides and fungicides next to neonicotinoid seed-treated maize fields that might have been applied to the maize or other crops in previous years.

Pesticide residue levels in wild plant pollen and nectar are also applicable to the exposure assessment of other FVI groups (e.g. lepidopterans, flies and beetles).

Table 24: Neonicotinoid residues detected in different matrices of wild plants adjacent to treated fields.

Adjacent crop	Wild plant	Matrix	Sample dates	Mean dose (ng/g)				Reference
				THX	CLO	IMI	THC	
Oilseed rape	54 plant species from field margins and hedges	Pollen	May-June 2013	14.81	NA	0.56	<0.04	Botías et al. (2015)
Wheat	54 plant species from field margins and hedges	Pollen	May-June 2013	0.14	NA	<0.16	<0.04	Botías et al. (2015)
Oilseed rape	54 plant species from field margins and hedges	Nectar	May-June 2013	0.10	NA	NA	NA	Botías et al. (2015)
Wheat	54 plant species from field margins and hedges	Nectar	May-June 2013	<0.10	NA	NA	NA	Botías et al. (2015)
Maize	Mustard, buckwheat, phacelia	Nectar*	Summer 2014 and 2015	NA	0.2-1.5	NA	NA	Mogren and Lundgren (2016)
Maize	Milkweed	Foliage	June 2014	NA	0.40	NA	NA	Pecenka and Lundgren (2015)
Maize	Milkweed	Foliage	July 2014 (1 month after planting)	NA	0.69	NA	NA	Pecenka & Lundgren (2015)
Oilseed rape	45 plant species from field margins and hedges	Foliage	May-June 2013	8.71	0.51	1.19	NA	Botías et al. (2016)
Maize	Buckwheat, mustard, partridge, pea, phacelia, *Plains coreopsis*, safflower, sunflower	Foliage	Summer 2014 and 2015	NA	0.5-13.5**	NA	NA	Mogren & Lundgren (2016)
Maize	Dandelion	Complete flower	Summer 2011	1.15	3.75	NA	NA	Krupke et al. (2012)
Various	Numerous plant species (20 m mean distance from field)	Complete flower	Summer 2012	7.20	1.40	1.10	NA	Stewart et al. (2014)
Oilseed rape	Field border plants	Complete flowers and foliage	April-May 2013 (2 days after sowing)	NA	1.20	NA	NA	Rundlöf et al. (2015)
Oilseed rape	Field border plants	Complete flowers and foliage	April-June 2013 (2 weeks after sowing)	NA	1.00	NA	NA	Rundlöf et al. (2015)

Table adapted from Wood & Goulson (2017). THX – thiamethoxam, CLO – clothianidin, IMI – imidacloprid, THC – thiacloprid. * Mogren and Lundgren (2016) obtained nectar from honey bees foraging on flowering wild plants. ** Range of concentrations, data on mean concentrations not available.

Table 25: Field studies that investigated neonicotinoid residues in bee-collected pollen and pollen sources adjacent to treated fields.

Species	Adjacent agriculture	Fraction of collected wildflower pollen (%)	Pesticide	Mean dose (ng/g)	Dose range	LOQ	LOD	Reference
Apis mellifera	Maize	95.3	Clothianidin	0.66	0.6-9.4	NA	0.1	Long & Krupke (2016)
Apis mellifera	Maize	73.7	Clothianidin	27.00	10.0-41.0	3.0	1.0	Pohorecka et al. (2013)
Apis mellifera	Maize	55.5	Clothianidin	9.71	1.0-88.0	NA	1.0	Krupke et al. (2012)
Apis mellifera	Oilseed rape	37.9	Clothianidin	13.90	6.6-23.0	1.5	0.5	Rundlöf et al. (2015)
Apis mellifera	Maize (colony inside field)	9.0 to 45.2	Clothianidin	0.84	NA	0.5	0.4	Cutler et al. (2014)
Apis mellifera	Oilseed rape (after bloom)	100.0	Thiamethoxam	0.12	0.1-0.3	0.4	0.1	Botías et al. (2015)
Apis mellifera	Oilseed rape (during bloom)	90.1	Thiamethoxam	0.20	0.1-1.8	0.4	0.1	Botías et al. (2015)
Bombus impatiens	Maize	99.4	Clothianidin	0.40	NA	0.5	0.1	Cutler & Scott-Dupree (2014)
Bombus impatiens	Maize	99.4	Thiamethoxam	0.10	NA	0.5	0.1	Cutler & Scott-Dupree (2014)

Table adapted from Wood & Goulson (2017). Of the detected pesticides only those are shown that were applied to the adjacent crop using seed treatment. Doses reported as <LOD or >LOD were standardised as equal LOD, doses reported as <LOQ as equal LOQ and doses reported as <"value" as equal to this value.

5.2.2.3 Food uptake

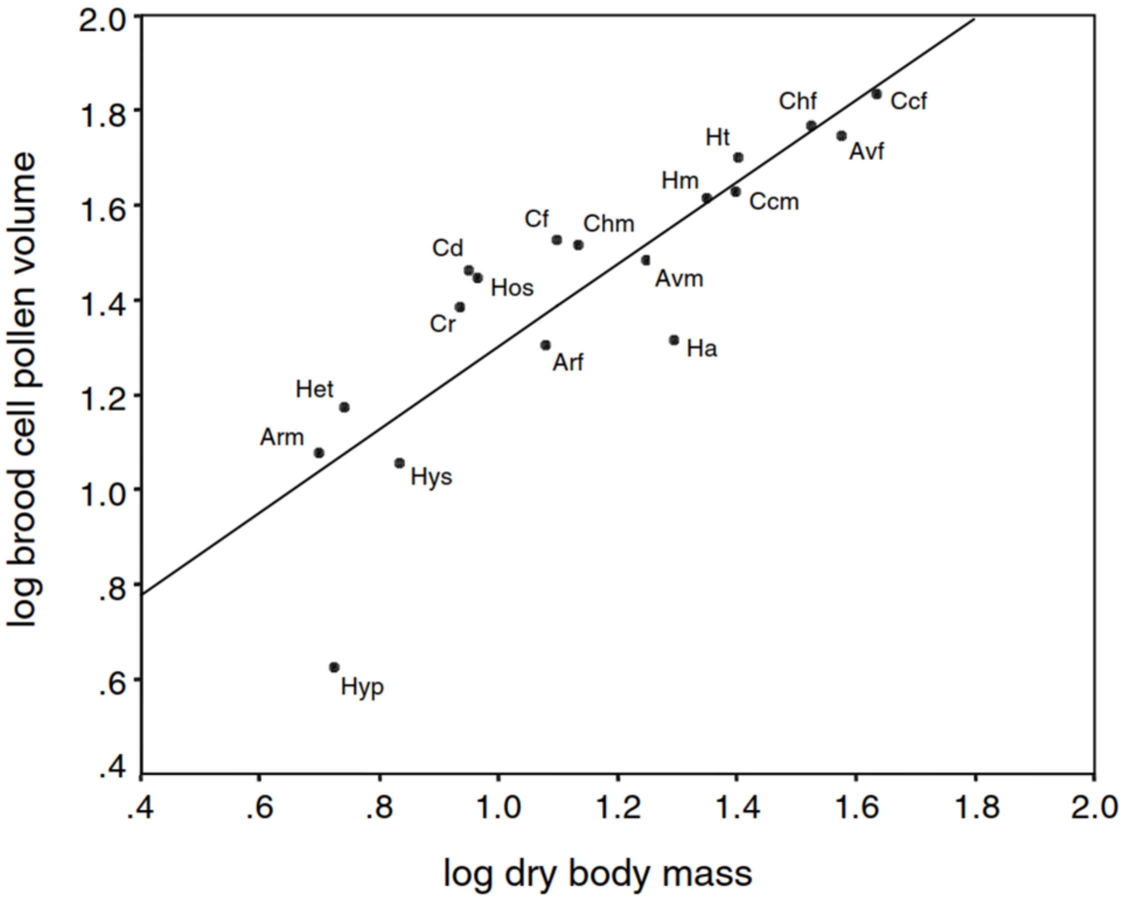
To assess individual exposure of bees it is necessary to quantify their food intake. In bees the fully developed adults usually procure their energy from carbohydrate-rich nectar, whereas larvae feed on pollen provision/pollen bread, a mixture of mostly protein-rich pollen and minor nectar content in ratios varying by species (Westrich 1990). Uptake of pesticides through food is an especially important pathway for sensitive life stages (e.g. larval stages) or queen bees in social species.

The amount of pollen that is consumed by larvae varies between species. Müller et al. (2006) investigated pollen provision volume in 14 oligolectic European bee species. Moreover, they then extrapolated pollen provision amount to other 35 bee species using dry body weight as a predictor (Figure 16). Their results show that larvae of bigger bee species need more pollen for their development. For the extrapolated bee species the estimated pollen provision volume was c. 5-100 mm^3 which translates to c. 1-1000 flower heads visited (of their specific host plant) or c. 0.1-20 plants visited. When extrapolating to the dry weight range of all European bee species (c. 1-210 mg) provision volumes of 2.7-280 mm^3 are estimated (values were calculated with data from European bee trait database (Roberts et al. unpublished) using the regression formula from Müller et al. (2006)). However, pesticide residues in pollen are usually only stated on a per mass basis. Pollen provision mass is a bad predictor of ingested amount of pollen since pollen-nectar ratios vary widely among bee species (Westrich 1990; Müller et al. 2006). To get a general idea: *Osmia bicornis* larvae consume around 100-250 mg (males) or 250-400 mg (females) of pollen provision at an ambient temperature of 20°C during their entire development (Radmacher & Strohm 2010). For an adequate assessment of larval food residue, exposure studies should include pollen counts. Pollen count could be easily converted to pollen mass if the mean mass of a single pollen grain is determined. Furthermore, nectar content of pollen provisions should also be investigated as an exposure source. Since polylectic bee species collect pollen from a wide variety of plant species (e.g. Coudrain et al. 2015, Sickel et al. 2015), their larval pesticide uptake is highly dependent on the proportion of contaminated pollen within the pollen provisions.

Bumble bee queens actively forage after hatching to support the first brood of workers. In this critical period *Bombus lucorum* queens gain about 100 mg of weight within 18 d by consuming pollen and nectar (Stoner 2016). Through the life stages of a bumble bee queen food consumption is variable in amount and composition. This complex change in diet needs further investigation to accurately evaluate pesticide exposure through food for queens.

To adequately assess pesticide exposure of other FVI groups (e.g. lepidopterans, flies and beetles) from nectar and pollen feeding, field-adjacent food uptake rates of their relevant life stages should be collected.

Figure 16: Relationship between the average brood cell pollen volume and the average dry body mass of the 14 bee species.



Footnote: Linear regression log y = 0.868 log x + 0.433 (F = 45.49, df = 17, $p < 0.001$, $R^2 = 0.74$). Arf - *Andrena ruficrus* (female), Arm - *Andrena ruficrus* (male), Avf - *Andrena vaga* (female), Avm - *Andrena vaga* (male), Ccf - *Colletes cunicularius* (female), Ccm - *Colletes cunicularius* (male), Cd - *Colletes daviesanus*, Chf - *Colletes hederae* (female), Chm - *Colletes hederae* (male), Cf - *Chelostoma florisomne*, Cr - *Chelostoma rapunculi*, Ha - *Hoplitis adunca*, Hm - *Hoplitis mocsaryi*, Ht - *Hoplitis tridentata*, Het - *Heriades truncorum*, Hos - *Hoplosmia spinulosa*, Hyp - *Hylaeus punctulatissimus*, Hys – *Hylaeus signatus*.

Source: Müller et al. (2006)

5.2.3 Soil

The majority of European bees species nest in the soil either by actively burrowing nests or using existing cavities (see p. 94 Nesting). Therefore, pesticide exposure by soil contact may be an important exposure pathway. Soil exposure may also be relevant for soil-dwelling life stages of other FVI groups (e.g. fly and beetle larvae).

Systemic pesticides are usually applied directly to the soil to be taken up by crops (e.g. seed treatment, granules). However, only a fraction of the applied pesticide load enters the plant body. Sur & Stork (2003) found that in crops such as maize, rice or cotton 1.6 to 20% of imidacloprid reach the plant, whereas the residual amount remains in the soil. In another study with maize Alford & Krupke (2017) could only detect a maximum amount of 1.34% of clothianidin applied as seed treatment in plant tissue. There have been several studies which measured neonicotinoid residues in agricultural soils (Table 26). These studies have shown a range of 0.4-13.3 ng/g for imidacloprid, clothianidin or thiamethoxam levels in different cultures with varying previous crops.

To assess the pesticide exposure of FVIs from soil it is not only important to know (peak) concentrations but also the persistence in the soil matrix. Goulson (2013a) reviewed half-lifes of neonicotinoid insecticides. His analysis showed that DT50 values vary substantially for imidacloprid (200 to >1000 d), thiamethoxam (7 to 353 d), clothianidin (148 to 6931 d), thiacloprid (3-74 d) and acetamiprid (31-450 d). Half-lives range from several days to years. Values over one year suggest possible accumulation or exposure by application of previous years even if these substances are not applied anymore. In an analysis of farmland soil samples Bonmation et al. (2005) found that imidacloprid doses were higher in soil that had been already treated the year before which may be the result of accumulation. Furthermore, in a reanalysis of two multi-year studies in seed-treated barley by Placke (1998b) and spray applications to orchard soils by Placke (1998a), Goulson (2013a) found evidence of neonicotinoid accumulation. Moreover, persistent substances can still be present even if they were not applied in the present season. Jones et al. (2014) investigated neonicotinoid residues in arable soils in multiple locations in England. They detected clothianidin and thiamethoxam in several fields where these insecticides had not been applied since three years ago.

A recently published study measured currently used pesticide content in 75 agricultural soils in the Czech Republic several months after the last pesticide application (Hvězdová et al. 2018). The soils contained mixtures of pesticide residues (51% soils with ≥5 pesticides) and levels were also noticeable (36% soils with ≥3 pesticides exceeding the threshold of 0.01 ng/g). The study shows that our knowledge of pesticide presence in agricultural soils is very limited and this is even more the case for soils in off-field habitats. Furthermore, it is difficult to link soil pesticide exposure to FVI exposure through soil contact. Pesticides can be sorbed to the soil and become bound residues with decreased bioavailability and degradation rates. This occurs especially in hydrophobic chemicals such as polycyclic aromatic hydrocarbons (Gevao et al. 2000; Semple et al. 2003). Water soluble compounds such as neonicotinoids might not be so prone to sorption and therefore retain their bioavailability to a greater extent. There is a need to investigate and quantify FVI exposure through soil contact.

Table 26: Neonicotinoid doses in agricultural soils published in recent studies.

Country	Year(s)	Samples collected	Previous crops	Mean neonicotinoid dose (ng/g)			Reference
				IMI	CLO	THX	
USA	2012	Spring, pre-planting	Various	4.00	3.40	2.30	Stewart et al. (2014)
UK	2013	Spring	Various	1.62	4.89	0.40	Jones et al. (2014)
Canada	2013 and 2014	Spring, pre-planting	Maize	NA	3.45	0.91	Limay-Rios et al. (2015)
UK	2013	Summer, with crop (10 months post planting)	Oilseed rape	3.03	13.28	3.46	Botías et al. (2015)
USA	2011 to 2013	Continuously	Maize and soybean	NA	2.0-11.2	NA	de Perre et al. (2015)
USA	2012 and 2013	Summer, with crop	Maize	NA	7.00	NA	Xu et al. (2016)
Canada	2012 to 2014	Summer, with crop	Oilseed rape	NA	5.70	NA	Xu et al. (2016)
Germany	2013	Autumn, pre-planting	Various	NA	2.10	NA	Heimbach et al. (2016)

Table adapted from Wood & Goulson (2017). THX – thiamethoxam, CLO – clothianidin, IMI – imidacloprid.

5.2.4 Stem/leaves

Systemic pesticides are designed to be taken up by crops from the soil: Of the applied amount of neonicotinoids 1.6-20% are actually absorbed by crops (Sur & Stork 2003). A study by Alford & Krupke (2017) showed an even lower uptake of maximum 1.34% in seed-treated maize. However, wild plants can also take up systemic pesticides. In the case of neonicotinoids a couple of studies have shown residues in wild plant stem or leaves (Table 24). Botías et al. (2016) investigated 45 wild plant species in field margins next to treated oilseed rape and found measurable doses of thiamethoxam, clothianidin and imidacloprid (8.71, 0.51 & 1.19 ng/g, respectively). In another study clothianidin levels in milkweed *Asclepias syriaca* in clothianidin-treated maize field margins were investigated (Pecenka & Lundgren 2015). Mean doses of 0.4 ng/g and 0.69 ng/g were detected around the time of maize planting and one month after. Mogren & Lundgren (2016) analysed foliage of seven wildflower species from pollinator strips adjacent to maize fields and detected a range of clothianidin doses from 0.5 to 13.5 ng/g. Sunflowers *Helianthus annuus* took up the highest levels of maximum, followed by buckwheat *Fagopyrum esculentum* and phacelia *Phacelia tanacetifolia*. Aside from FVI herbivore life stages even other FVIs might be exposed to pesticide residues by contact to plant stem or leaves. Neonicotinoids and other non-systemic pesticides are applied in crops when wild bee species such as leaf cutter bees (*Megachile* ssp.; see p. 94 Nesting) are collecting nesting materials. Generally, FVI species might be exposed to pesticide residues in or on plant material (e.g. herbivore life stages of lepidopterans and beetles or bees collecting nest materials).

5.2.5 Non-nectar fluids

Flower-visiting insects may also be exposed to pesticides when consuming water. Fletcher & Barnett (2003) reported an incident where honey bees were poisoned from drinking water during dry weather conditions. They drank from puddles on a field that was previously applied with the herbicide paraquat. Samson-Robert (2014) investigated if puddles in maize fields are contaminated with neonicotinoids. Amongst others they found clothianidin and thiamethoxam in concentrations from

0.1 to 55.7 ng/mL and 0.1 to 63.4 ng/mL, respectively. Considering daily water intake they concluded that maximum concentrations of both insecticides might represent a risk to honey bees. In another study maximum total concentrations of both substances were detected at 17.83 ng/mL in puddles and ditches around maize fields (Schaafsma et al. 2015). Lepidopteran species exhibit a behaviour termed “mud puddling” where they take up fluids out of puddles, dung or carrion to ingest nutrients which might also expose them to pesticides residues containing in small ephemeral in- or off-field water bodies (Downes 1973; Adler 1982; Boggs & Jackson 1991).

Another water source for FVIs might be guttation droplets secreted by some plant species. Bonmatin et al. (2015) concluded that such an exposure does only occur at toxicologically relevant doses in crops treated with systemic pesticides. Spray treatments lead to doses that are lower by three orders of magnitude than when applying seed treatment or granules. Concentrations of systemic neonicotinoids in guttation fluid vary greatly (Tapparo et al. 2011) but can be up to four to five orders of magnitude higher than concentrations in nectar (Godfray et al. 2014). First evidence of high pesticide content was provided by Girolami et al. (2009). They found concentrations of thiamethoxam and clothianidin of up to 100 µg/mL and up to 200 µg/mL for imidacloprid in maize. In another maize study maximum residue concentrations of over 300 µg/mL imidacloprid and over 100 µg/mL thiamethoxam or clothianidin were detected (Tapparo et al. 2011). Reetz et al. (2016) measured thiamethoxam content of oilseed rape guttation droplets and residues in honey bee honey sacs. Guttation fluid contained of 70-130 ng/mL clothianidin (metabolite of thiamethoxam) at cotyledon stage. Honey sac concentrations of both, thiamethoxam and clothianidin, were only detected in 14% of all samples at levels between 0.1-0.95 ng/mL. However, the authors could not directly link the neonicotinoid exposure to guttation fluid since they might have also taken up pesticides from water bodies or nectar. A clear link of pesticide residues in guttation droplets and pesticide uptake of bees remains to be provided (Wood & Goulson 2017).

Open questions/research opportunities

- Study exposure of other organisms than the honey bee, especially other FVI groups.
- Establish link of habitat to FVI exposure for less well-researched matrices (soil, stem/leaves, guttation water, honeydew, extrafloral nectaries, puddles).

5.3 General overview of exposure scenarios

As outlined in chapter 4 Pesticide exposure of habitats and chapter 5 Pesticide exposure of flower-visiting insect species there are multiple sources and pathways of FVI pesticide exposure. For in-field exposure these pesticide sources and their subsequent pathways as well as the possibly exposed life stages of FVIs, are summarised in Table 27. In the case of off-field exposure, pesticide sources and their subsequent pathways, as well as the possibly exposed life stages of FVIs, are similar to in-field exposure (Table 27). However, the difference is that pesticides are not directly applied in these areas but are transported there by secondary exposure processes (field edge overspray, spray drift, dust drift, run-off).

Table 27: Overview of in-field exposure scenarios.

Exposure source		FVI groups
Spray application	**Solid application**	
Overspray (+ drift)	Dust dispersion	All FVI species
Flower deposit		All adult life stages of FVIs and nectar/pollen feeding larvae of bees and lepidopterans
Stem/leaf deposit		herbivore life stages of lepidopterans (caterpillars) and beetles and adult bees that collect plant materials for nest building
Soil deposit		Soil-dwelling fly and beetle larvae and soil-nesting or soil-collecting bees.
Systemic flower deposit		All adult life stages of FVIs and nectar/pollen feeding larvae of bees and lepidopterans
Systemic stem/leaf deposit		herbivore life stages of lepidopterans and beetles and adult bees that collect plant materials for nest building
Aqueous deposit		All FVI species

Spray drift is only a relevant exposure process in off-field areas.

5.4 Detailed description of exposure estimation

5.4.1 In-field habitats

5.4.1.1 Spray applications

Overspray

Relevant FVI groups: All FVI species

When pesticides are applied, FVIs that are situated in-field will be exposed. If the FVI is flying over the crop the pesticide dose that is received is relative to the FVI individual's surface. The application rate (AR) can be displayed on a per cm² basis (Eq. 1):

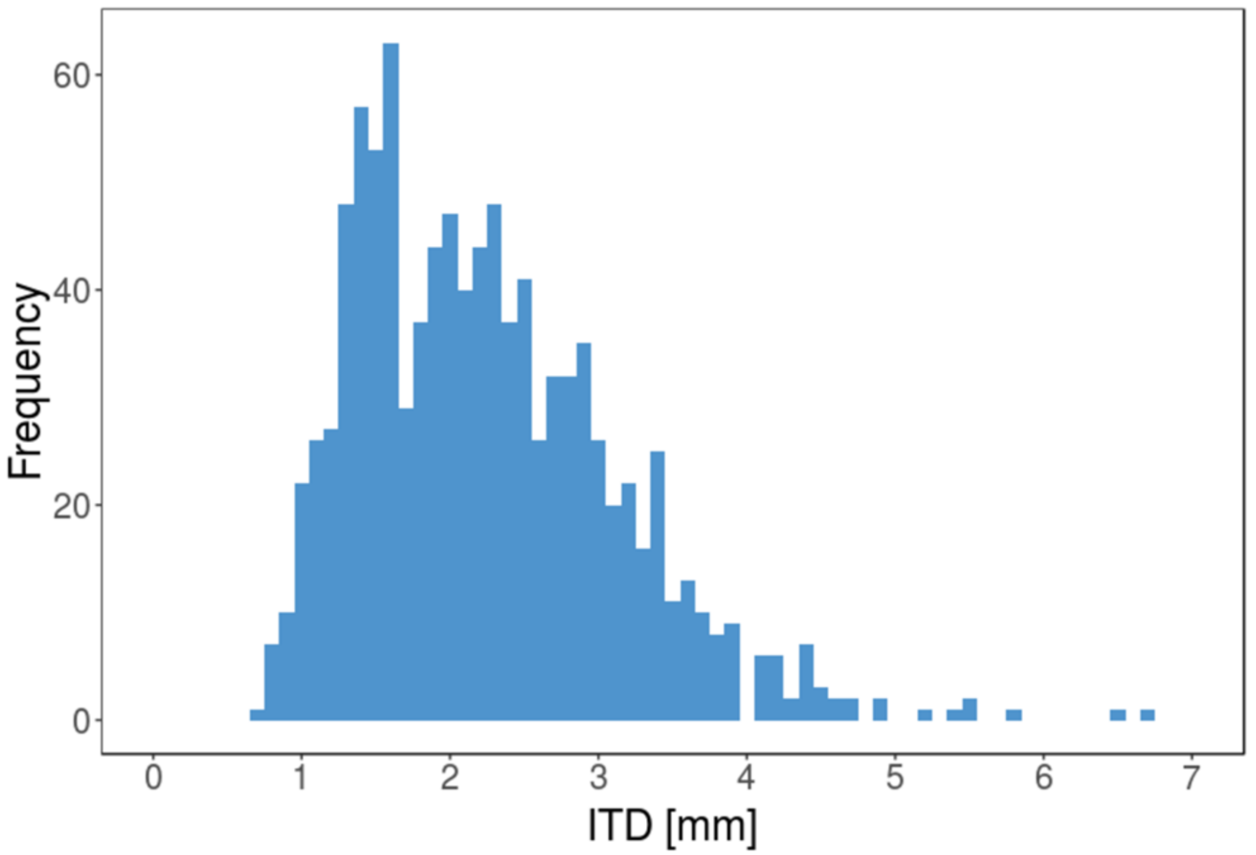
$$AR\ [g/ha] = 10 * AR\ [ng/cm^2] \qquad (1)$$

In the case of a downwards overspray (2D applications in arable crops) the individual dose (ID [ng/cm²]) that is applied can be estimated using Eq. 2. EFSA (2013) assumed the one-sided surface area ($A_{surface}$ [cm²]) of a honey bee to be 1 cm². Poquet et al. (2014) estimated the apparent exposure surface area as 1.05 ± 0.33 cm² (mean ± SD). A worst case scenario estimate of individual surface in this case would be 1.21 cm² (upper limit of 95% CI).

$$ID = 10 * AR\ [ng/cm^2] * \ A_{surface} \qquad (2)$$

Bee species vary considerably in size as evident by the range of their intertegular distances (ITD; the width of the body measured between the wing bases) (Figure 17). Therefore, the exposure surface area has to be determined for every species and the honey bee cannot be used as a surrogate. However, it might be possible to interpolate surface area values using the ITD. It is already known to be a good proxy for bee mass (Cane 1987) and foraging distance (Greenleaf et al. 2007). With a sufficient database of surface area measurements it is possible that a relationship of surface area and ITD could be established.

Figure 17: Distribution of ITDs among European bee species.



Footnote: Dataset includes 1003 out of 1936 species.

Source: own illustration. Data from European bee trait database (Roberts et al. unpublished)

FVIs might also be exposed to pesticides by sidewards or upwards overspray (3D applications in e.g. orchards, vineyards). For this case EFSA proposed to calculate the ID assuming half the exposure as in the downwards scenario (EFSA 2013). However, this assumption needs to be substantiated. Furthermore, another way for FVIs to get exposed by overspray is when sitting in flowers, on leaves or on the stem. Moreover, wing exposure (in-flight or stationary) is currently not included in exposure estimation (Poquet et al. 2015).

Since a specification of these different overspray scenarios might unnecessarily complicate the risk assessment process, we propose to conservatively assume an exposure of the total physical surface area (body and wings). This would incorporate all the named overspray exposure scenarios. Poquet et al. (2014) determined the honey bee physical surface to be 3.27 ± 0.23 cm^2 (mean ± SD). It is unclear if this approach incorporates in-flight exposure to the body and beating wings of a FVI. Poquet et al. (2015) estimated such an exposure using a modelling procedure. However, their findings need to be validated with experimental data possibly using a similar setup to Poquet et al. (2014). Data on overspray exposure of groups of FVIs other than bees are scarce and therefore, we recommend to use the conservative estimate of total physical surface area for all FVIs.

Flower deposit

Relevant FVI groups: All adult life stages of FVIs and nectar/pollen feeding larvae of bees and lepidopterans

Plant flowers are exposed when pesticides are applied in bloom. Therefore, FVIs that visit these flowers afterwards come in contact with contaminated nectar or pollen. Furthermore, species that actively feed on nectar or pollen as larvae or are being fed these substances as larvae are orally exposed.

Contact exposure

EFSA (2015) devised a framework to estimate in-field, on-crop exposure of NTAs which is applicable to this case with minor alterations. The pesticide mass that arrives at the crop canopy ($A_{d,p}$ [kg m^{-2}]) is estimated using the Leaf Area Index (LAI), the fraction of the applied mass that is intercepted by the crop canopy (f_i) and the application rate of the pesticide ($A_{d,f}$ [kg m^{-2}]) (Eq. 3).

$$A_{d,p} = \frac{1}{LAI} * f_i * A_{d,f} \qquad (3)$$

A tier 1 exposure assessment would consider a 100% intercept of the pesticide by the crop canopy (f_i = 1) and a total area of single-sided leaf that equals the surface area of the field (LAI = 1). LAI and f_i can be adapted to more realistic values in refined exposure assessment (e.g. base LAI on crop developmental stage). Furthermore, dissipation of the applied pesticide and wash-off can also be taken into account. The dissipation rate of the pesticide mass that has been adsorbed to the plants' surface (R_{dsp} [kg m^{-2} d^{-1}]) is estimated using the dissipation half-life ($DisT50_p$ [d]) and the pesticide mass that was intercepted by the crop canopy ($A_{d,p}$)(Eq. 4).

$$R_{dsp} = \frac{\ln(2)}{DisT50_p} * A_{d,p} \qquad (4)$$

Wash-off following precipitation can also be estimated: The rate of pesticide wash-off from the crop canopy ($R_{w,p}$ [kg m^{-2} d^{-1}]) is calculated using multiple parameters such as precipitation (P [m d^{-1}]), an empirical wash-off factor (w_p [m^{-1}]), the extinction coefficient for diffuse solar radiation (κ) and an empirical parameter a. (Eq. 5) However, wash-off should be included only with great care in exposure calculations since pesticide applications are usually performed during sunny weather to avoid pesticide loss on crop plants. Therefore, wash-off should generally not be considered in realistic worst-case scenarios. All these parameters can also be adapted for refined assessment. For further details please see EFSA (2015).

$$R_{w,p} = (1 - e^{-P*w_p}) * \left(1 - e^{-\kappa*LAI} * P - \left(a * LAI * \left[1 - \frac{1}{1 + (1 - e^{-\kappa*LAI}) * P/a * LAI}\right]\right)\right) * A_{d,p} \qquad (5)$$

We propose that this or similar frameworks can be easily adapted to assess flower deposits after spray application in defining a Flower Area Index (FAI [m^2 m^{-2}]) that incorporates the crop-specific fraction of flower-to-leaf area ($f_{f,l}$) (Eq. 6). Such an index would need to be validated by field data.

$$FAI = f_{f,l} * LAI \qquad (6)$$

Oral exposure

This exposure can be estimated similar to oral exposure from systemic pesticides (section Systemic flower deposit, p. 115). However, it should be taken into account that topical plant residues might behave differently in terms of temporal stability, i.e. chemical breakdown due to weather, and physical dissipation.

Stem/leaf deposit

Relevant FVI groups: herbivore life stages of lepidopterans and beetles and adult bees that collect plant materials for nest building

Contact exposure

This assessment can be conducted as proposed for NTAs by EFSA (2015). Please see the previous section (Flower deposit, p. 113).

Oral exposure

This exposure can be estimated similar to oral exposure from systemic pesticides (section Systemic flower deposit, p. 115). However, it should be taken into account that topical plant residues might behave differently in terms of temporal stability, i.e. chemical breakdown due to weather, and physical dissipation.

Soil deposit

Relevant groups: Soil-dwelling fly and beetle larvae and soil-nesting or soil-collecting bees

A certain amount of an applied pesticide will reach the in-field soil. Flower-visiting insects that dwell or nest in the soil will be subsequently exposed. EFSA (2015) proposed the following estimation procedure. The pesticide mass that is not intercepted by the plant canopy or is washed off from it and therefore reaches the soil surface ($A_{d,s}$ [kg m^{-2}]) can be estimated using Eq. 7: Parameters in this equation are the application rate (A_f [kg m^{-2}]), the fraction of the applied mass that is intercepted (f_i) and the fraction that is washed off from the plant canopy (f_w).

$$A_{d,s} = \left((1 - f_i) + f_i * f_w\right) * A_f \qquad (7)$$

It is further possible to calculate the amount of remaining pesticide residue in the soil ($A_{plateau}$ [kg m^{-2}]) right before an application after a hypothetical infinite number of annual applications when incorporating the following variables (Eq. 8): the averaging depth of interest (z_{avg} [m]), the plough depth (z_{til} [m]), the time between annual applications (t_{cycle} [d]), the first-order degradation rate coefficient (k_{ref}) and a factor describing the effect of soil temperature on k_{ref} (f_T).

$$A_{plateau} = \frac{Z_{avg}}{Z_{til}} * A_{d,s} * \frac{e^{-t_{cycle} * f_T * k_{ref}}}{1 - e^{-t_{cycle} * f_T * k_{ref}}} \qquad (8)$$

A maximum pesticide mass in the soil (A_{peak} [kg m^{-2}]) is then estimated by summing up the applied pesticide mass and remaining pesticide residues (Eq. 9).

$$A_{peak} = A_{plateau} + A_{d,s} \qquad (9)$$

Tier 1 exposure assessment assumes that the pesticide that reaches the soil equals the application rate. Total applied mass is calculated as the sum of all applications within a growing season. In refined exposure assessment, models and their parameters may be adapted for more realistic settings. For further details please see EFSA (2015) & EFSA PPR Panel (2017).

It has to be noted that soil residues of pesticides are not quantitatively equivalent to FVI residues after soil contact (e.g. while digging a nest or collecting mud for brood cell construction). However, there is currently no applicable framework to estimate pesticide transmission from soil to FVI.

Systemic flower deposit

Relevant groups: All adult life stages of FVIs and nectar/pollen feeding larvae of bees and lepidopterans

The framework to assess oral exposure of FVIs to systemic pesticides by consumption of nectar or pollen is adapted from the suggestions in EFSA (2015). Extrafloral nectaries and honeydew can also be addressed within this framework. The oral intake of a specific pesticide (intakeFVI [g]) is calculated as the sum of pesticide intake from all of food items (n) that are consumed by an FVI species. The intake per food item is defined as the product of the fraction of this food item (ffi) in the diet, the pesticide content of that food item ($content_i$ [ng g^{-1}]) and the daily food intake by the FVI (DFI_{FVI} [g]) over a specific time period (Eq. 10).

$$intake_{FVI} = \sum_{i=1}^{n} ff_i * DFI_{FVI} \qquad (10)$$

EFSA (2015) further suggested how to estimate the food intake. They proposed obtaining the DFI from allometric equations that relate energy requirement to body mass. It is unlikely that such an approach is feasible for FVIs: They do not only visit flowers to collect the energy-rich nectar but also to gather nutrient-rich pollen. For bumble bees there is evidence that adults can assess the nutrient content and quality of pollen by taste (Ruedenauer et al. 2015). They further seem to prefer visiting flowers where they can gather pollen with a high protein content as food for their larvae (Hanley et al. 2008). Therefore, bee larvae oral exposure needs a different approach based on their protein demand. Moreover, an adult bumble bee collecting pollen is also likely to collect nectar thus potentially exposing it orally to a pesticide. Other FVIs such as pollen beetles (e.g. from the families Melyridae, Nitidulidae and Oedemeridae) feed exclusively on pollen. The specific diet of FVIs throughout their life cycle has to been considered for exposure assessment. EFSA (2013) did already collect some data on adult and larval bees' consumption of nectar (sugar solution as proxy) and pollen (Table 28). However, data are still scarce and there is an evident need to gather more knowledge on FVIs' diet before incorporating it into quantitative exposure assessment. Meanwhile, a conservative approach would be the assumption that FVIs are only consuming one single contaminated food item (EFSA 2015).

The pesticide content of specific food items can be deduced from its RUD (residue unit dose). This is the concentration in/on nectar or pollen calculated for an application rate of 1 kg a.i./ha or 1 mg a.i./seed after a specific time. RUDs have to be determined through chemical analysis of the relevant plant compartment after pesticide application. This was already done for some pesticide crop combinations using multiple application methods (EFSA 2008, 2013).

Table 28: Sugar (nectar surrogate) and pollen consumption of different bee species types and their larvae.

Species type	Adult consumption [mg/bee/day]		Larval consumption [mg/bee/day]	
	Sugar	Pollen	Sugar	Pollen
Honey bee forager	32.0-128.0	0.0	11.9	0.3-0.4
Honey bee nurse	34.0-50.0	6.5-12.0		
Bumble bee	73.0-149.0	26.6-30.3	23.8	10.3-39.5
Solitary bee	18.0-77.0	10.2	1.8	12.9

Adapted from EFSA (2013).

Exposure assessment should explicitly consider the feeding of larvae, as a susceptible life stage, with contaminated pollen or nectar. Therefore, larval diet should be investigated as well as the imago's. In the case of bees it should be distinguished with great care between nectar and pollen consumption since the pollen-nectar ratio of larval provision varies considerably between bee species. Counting pollen grains in the so-called bee bread (i.e. pollen provision) and converting that figure into mass (e.g. by multiplying with mean mass of a single pollen grain) is more appropriate than just weighing the pollen provision (Müller et al. 2006).

Systemic stem/leaf deposit

Relevant groups: herbivore life stages of lepidopterans and beetles, adult bees that collect plant materials for nest building

This assessment can be conducted as outlined in the previous section (section Systemic flower deposit, p. 115).

Aqueous deposit

Relevant groups: All FVI species

Guttation water

Several crops can exude water on the tips and edges of leaves that may contain very high pesticide concentrations (EFSA PPR Panel 2012). Therefore, EFSA (2013) proposed an exposure assessment for this case that is applicable to FVIs. The individual dose (ID [µg]) can be estimated from predicted environmental concentrations in the guttation water (PEC [µg µL-1]) and the water consumption (W [µL]) of an FVI (Eq. 11).

$$ID = W * PEC \tag{11}$$

In lower tier acute risk assessment the PEC is assumed to be 100% of the water solubility of the pesticide. The water consumption of a honey bee was estimated to be 11.4 µL/d. The parameters can be set to more realistic values in higher tier assessment. For further details please see EFSA (2013). The water consumption value would have to be adapted to other FVI since it is unclear if the honey bee value is protective. In an US EPA whitepaper their daily water consumption was estimated as 0.45 – 1.8 mL based on honey bee observations or 47 µL based on direct measurements of water requirements of the brown paper wasp *Polistes fuscatus* (a similar surrogate) (Environmental Fate and Effects Division (EFED) et al. 2012). Further research is needed to gain a more reliable estimate of wild bees and other FVIs water consumption (e.g. mud puddling in lepidopterans). Moreover, it should be investigated to what extent FVI use guttation droplet as a water source.

Puddles

Puddle water can be an important source of pesticide exposure for FVIs since these ephemeral water bodies can contain numerous pesticides (Samson-Robert et al. 2014). EFSA (2012) recommended that its pesticide content should be estimated from run-off water concentrations using FOCUS run-off scenarios. Please see this document for further details and EFSA (2013).

5.4.1.2 Solid application

Soil deposit

Relevant groups: Soil-dwelling fly and beetle larvae and soil-nesting or soil-collecting bees

A guidance document for seed treatment was drafted 2014 to evaluate exposure from solid applications to the environmental surroundings (SANCO 2014). For the soil exposure assessment, it was proposed to use a dual approach. When small seeds (diameter <0.5 cm) are sown, soil load can be calculated similar to spray applications. The dose of active substance can be divided by the mass of the upper 5 cm of the soil to derive a predicted initial environmental dose (PIED) as a measure for acute exposure of soil organisms. For seeds larger than 0.5 cm in diameter which are sown at lower densities and a depth of 5 cm an alternative scenario is proposed. In this approach it is assumed that treated seeds have a spherical shape and a bigger sphere around them is exposed. Therefore, a PIED can be calculated by the dose of active substance divided by the soil mass within this sphere of influence. For further details please see SANCO (2014). Furthermore, there will be dust depositions to the soil surface (see off-field Soil deposit, p. 120).

However, it has to be acknowledged that the discrimination of smaller and bigger seeds as well as the sphere calculation approach for bigger seeds are quite theoretical and may use arbitrary assumptions. Depending on seed density influence spheres might overlap which could lead to areas of increased residues. Seed depths lower than 5 cm might also cause higher local residues. Furthermore, the overall approach does not include in-soil chemical/microbial degradation, transport processes or the formation of bound residues which are soil and pesticide-specific properties. Moreover, exposure of deeper soil layers is not considered which might be relevant for soil-dwelling life stages of FVIs. A suitable modelling approach should be developed to allow for a more precise and scientifically sound soil exposure assessment. Furthermore, systemic loads in soil may be taken up by wild plants and subsequently expose FVI life stages through plant matrices (e.g. nectar/pollen, stem/leaves).

As it is the case with soil residues following spray applications, it has to be noted that soil residues of pesticides are not quantitatively equivalent to FVI residues after soil contact (e.g. while digging a nest or collecting mud for brood cell construction). However, there is currently no applicable framework to estimate pesticide transmission from soil to FVI.

Aqueous deposit

Relevant groups: All FVI species

Puddles

In the framework of the SANCO draft seed treatment guidance document no in-field recommendations for the estimation of dust exposure to puddles are provided (SANCO 2014). However, these small ephemeral water bodies may occur in freshly drilled soil. Off-field puddle scenarios from the seed treatment guidance document may be used to evaluate exposure (see p. 121 Aqueous deposit).

Remaining exposure scenarios

All remaining solid application exposure scenarios (Table 27) can also be implemented as in the systemic spray application exposure assessment framework as long as the necessary plant uptake data is provided. In the meantime, we recommend that pesticide manufacturers provide RUDs for contact and oral exposure assessment in all relevant matrices (e.g. nectar/pollen, stem/leaf) gathered from field trials.

5.4.1.3 Further exposure sources

Stem application

An application technique that originated in forestry is stem application or stem injection. In this procedure a hole is drilled into the stem of a tree and the systemic pesticide in applied directly into the xylem (Helson et al. 2001). This technique is also feasible for viticulture (Düker & Kubiak 2015). However, it is not well-established in Europe, yet, and therefore not considered in pesticide risk assessment.

The following section of the framework above would apply to this application technique:

Spray applications (p. 111)

- Systemic flower deposit
- Systemic stem/leaf deposit

Irrigation

Pesticides may also be applied by irrigation (e.g. Miorini et al. 2017). This application technique is also not considered in risk assessment.

The following section of the framework above would apply to this application technique:

Spray applications (p. 111))

- Flower deposit
- Stem/leaf deposit
- Soil deposit
- Systemic flower deposit
- Systemic stem/leaf deposit

5.4.2 Off-field habitats

5.4.2.1 Spray applications

Overspray and drift

Relevant groups: All FVI species

General considerations

Spray drift is a major source of pesticide influx into non-target habitats. The deposited fraction (f_d) of the application rate can be expressed as a function of the distance to the field edge (x [m]) and crop-specific parameters (a, b) that have to be evaluated for different cropping systems (e.g. field culture, orchard, vineyard, hops) and cultures (e.g. oilseed rape, wheat, strawberry) (Eq. 12). These resulting values from field trials were summarised in drift deposition tables (Rautmann et al. 2001).

$$f_d = a * x^b \tag{12}$$

However, these tables are only available for German crops and therefore only valid for Central Europe. EFSA (2013) called for a harmonisation and improvement of the database of drift deposition estimates in the EU. For further details see (EFSA 2013).

Contact exposure by spray drift

The deposited pesticide fraction can be used together with the application rate (AR) and the exposed surface area ($A_{surface}$ [cm^2]) to assess direct exposure (individual dose: ID [mg/cm^2]) by contact with spray drift (Eq. 13). It needs to be surveyed if the exposed surface area has to be adapted to account for the manner in which the FVI is encased by the drift cloud since this exposure differs from direct overspray (e.g. from half to the entire FVI surface).

$$ID = f_d * 10 * AR[ng/cm^2] * A_{surface} \tag{13}$$

First meter of a field margin

A special case is the first meter of a field margin adjacent to a field culture. Usually a significant part of the field margin is oversprayed to achieve 100% application even at the field edge. It was conservatively estimated that this first meter is applied with 30% of the application rate which should be then assumed instead of the drift deposition factor (f_d) (Schmitz et al. 2013; Hahn et al. 2015a).

Aqueous deposit

Relevant groups: All FVI species

Surface water

Pesticide concentrations in surface water may be calculated using the FOCUS model framework. However, it was suspected that these concentrations are usually too low to cause any harmful effects in bees (EFSA 2013). This claim would need to be substantiated for bees and other FVI groups.

Remaining spray exposure scenarios

Using the spray drift deposition factor and FOCUS run-off water scenarios the remaining exposure assessment following spray application can be conducted using the in-field framework (see In-field habitats, p. 111; e.g. Overspray (p. 111), Flower deposit (p. 113), Stem/leaf deposit (p. 114), Soil deposit (p. 114), Systemic flower deposit (p. 115), Systemic stem/leaf deposit (p. 116) and Aqueous deposit (p. 116)).

5.4.2.2 Solid applications

Dust dispersion

Relevant groups: All FVI species

General considerations

When planting pesticide-treated seeds or applying granules, there can be pesticide dust emissions that may be transported to plants, soil and FVI species. The amount of dust from planting pesticide-treated seeds is highly variable and depends mainly on drilling equipment and the seed quality. For drilling equipment, the general rule is that mechanic devices produce less dust drift than pneumatic machines. Further details can be found in SANCO (2014). Seed quality parameters are expressed in a proxy parameter "Heubach a.s." as the amount of active substance measured with the Heubach method released with dust per sown area (Eq. 14). This value can be used in the determination of predicted environmental concentrations.

$$Heubach\ a.s. [g\ a.s.\ in\ dust/ha] = \frac{Heubach\ [g\ dust/ha] * a.s. [\%\ in\ dust]}{100} \tag{14}$$

Heubach values have been determined for several crops (see SANCO (2014) for further details).

Dust drift contact exposure

EFSA (2015) proposed to include all relevant exposure routes into a more general modelling approach. It seems rather unproblematic to incorporate contact exposure by dust drift into overspray scenarios (Overspray, p. 111) using Heubach a.s. values.

Soil deposit

Relevant groups: Soil-dwelling fly and beetle larvae and soil-nesting or soil-collecting bees

To estimate soil surface deposition of dust drift SANCO (2014) proposed a scheme that incorporates the first meter beyond the field edge. In this framework soil is regarded as a two-dimensional space where dust is deposited on. Using the Heubach a.s. parameter and a crop-specific deposition factor, a $PEC_{2D\ dust\ ground\ deposition}$ [g a.s./ha] can be calculated (Eq. 15). Further details such as deposition factor are available in SANCO (2014).

$$PEC_{2D\ dust\ ground\ deposition} = Heubach\ a.s. * deposition\ factor \tag{15}$$

However, exposure of deeper soil layers is not considered which might be relevant for soil-dwelling life stages of FVIs. Furthermore, systemic loads in soil may be taken up by wild plants and subsequently expose FVI life stages through plant matrices (e.g. nectar/pollen, stem/leaves).

EFSA (2013) suggested to develop physical models of dust deposition into field margins that incorporate wind speed and angle. Such approaches are not incorporated in the SANCO (2014) framework but might be added to existing models.

Flower deposit & stem/leaf deposit

Relevant groups: All adult life stages of FVIs and nectar/pollen feeding larvae of bees and lepidopterans as well as herbivore life stages of lepidopterans and beetles and adult bees that collect plant materials for nest building

To address dust deposition on three-dimensional objects (i.e. attractive wild plants), SANCO (2014) introduced an extrapolation factor to apply to two-dimensional deposit estimates (see Soil deposit, p. 120). Using the $PEC_{2D\ dust\ deposition}$ value and this 3D extrapolation factor a $PEC_{3D\ dust\ deposition}$ [g a.s./ha] can be calculated (Eq. 16). The extrapolation factor was set to a value of 13 by evaluating results from several dust field studies in different crops (see SANCO (2014) for further details).

$$PEC_{3D\ dust\ ground\ deposition} = PEC_{2D\ dust\ ground\ deposition} * 3D\ extrapolation\ factor \tag{16}$$

Aqueous deposit

Relevant groups: All FVI species

Surface water and puddles

Exposure of water bodies by seed treatment dust deposition is not addressed in current FOCUS surface water scenarios. Therefore, SANCO (2014) suggested to estimate the $PEC_{surface\ water\ dust}$ [µg a.s./L] using the $PEC_{2D\ dust\ ground\ deposition}$ and the relative volume of the water body [L/m^2] (Eq. 17). For further details please see SANCO (2014).

$$PEC_{surface\ water\ dust}\ [\mu g\ a.s./L] = \frac{PEC_{2D\ dust\ ground\ deposition}\ [g\ a.s./ha] * 100}{water\ volume\ [L/m^2]} \tag{17}$$

Remaining dust exposure scenarios

There are some aspects that are not explicitly addressed in the current guidance documents (EFSA 2013; SANCO 2014) and scientific opinions (EFSA 2015). Systemic deposits in wild plant matrices (e.g. nectar/pollen, stem/leaves) following dust drift or systemic run-off are not incorporated as well as dust deposits/run-off following granule application. However, these remaining solid application exposure scenarios (Table 27) may also be implemented similar to the in-field exposure assessment framework. This may be feasible as long as the necessary data to model exposure of the respective matrix (plant, soil) is provided. In the meantime, we recommend that pesticide manufacturers provide RUDs for contact and oral exposure assessment and all relevant matrices (e.g. nectar/pollen, stem/leaf, soil) that are gathered from field trials for these off-field scenarios following a solid application.

5.4.3 Landscape-scale exposure modelling

Since FVIs are mobile species, knowledge about their spatio-temporal exposure patterns is required for the assessment of possible populations effects. This was also recognised by EFSA in their NTA scientific opinion (EFSA 2015). Please see section 7.2 for a feasibility study on agent-based modelling of FVI species in the European landscape.

5.5 Conclusion

There is extensive evidence that bees are exposed to pesticides not only through direct overspray or spray/dust drift but also by consuming contaminated food item such as pollen and nectar or water. Furthermore, bees can be exposed by collected nesting materials or digging their nests in the soil. These exposure pathways are probably also valid for life stages of other FVIs who consume pollen or nectar (e.g. lepidopterans, beetles, flies), stem or leaf material (e.g. lepidopterans, beetles), water or nest in the soil (e.g. beetles, flies). There is some ecological trait information for bees which allows for evaluation of their exposure probability to specific habitat matrices. However, this database needs to be expanded for bees and established for other FVI groups. Furthermore, pesticide residue data in all relevant matrices needs to be collected (especially in off-field non-target plants) to quantitatively assess FVI exposure and create/validate adequate exposure models. Landscape-scale modelling can be a valuable tool to evaluate FVI exposure in space and time.

Open questions/research opportunities

- There is a need for landscape-scale exposure assessment to realistically evaluate FVI exposure by food or contact following spray and solid applications in in-field and off-field scenarios.
- A comprehensive database of residues in all relevant matrices (e.g. pollen/nectar, stem/leaves, soil, water) of crops and non-target areas should be collected.
- Studies are needed to determine dust dispersion after granule applications.
- Off-field pesticide loads in soil and plant matrices need to be investigated for various pesticide classes.

6 Flower-visiting insect sensitivity towards pesticides

Philipp Uhl, Carsten Brühl

6.1 Bees

6.1.1 Laboratory toxicity

6.1.1.1 Lethal effects

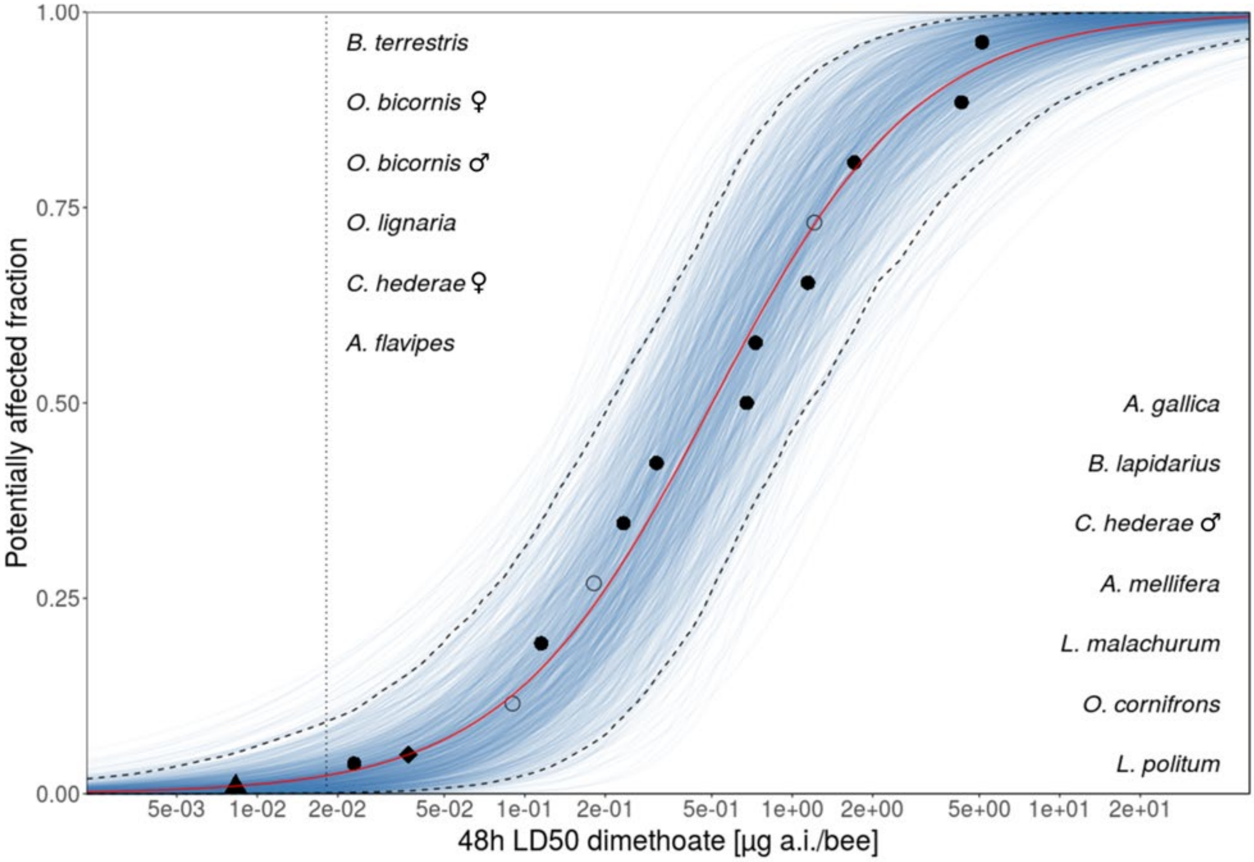
Since the honey bee is a test organism in European pesticides risk assessment there are acute toxicity data for all registered pesticides. However, other bee species' sensitivity towards pesticides is usually unknown which makes it difficult to establish the honey bee as a surrogate organism for risk assessment of wild bees or even other FVIs. The first acute effect studies on a European wild bee species were conducted by Elisabeth Ladurner 15 years ago, already proposing a method for an oral toxicity test (Ladurner et al. 2003, 2005). Arena & Sgolastra (2014) analysed the at that time available literature and found a bridging factor of 10 on top of a honey bee LD50 to cover wild bee species sensitivity in 95% of all cases. However, due to data shortage they also had to rely on data of tropical species (9 out of 19 species) which are therefore not representative for European wild bee fauna. Uhl et al. (2016) assessed acute toxicity of dimethoate, a commonly used reference substance in insect testing, towards five European wild bee species (Table 29) and generated a species sensitivity distribution (SSD) with the obtained data (Figure 18). In the meantime, additional experiments with other bee species were carried out and so in this report three species could be added to this database for a total of eight species. The lower 95% confidence limit of the hazardous dose (HD5) was similar to the honey bee 48 h LD50 when applying the bridging factor of 10 (Figure 18). This indicates that using the bridging factor would allow for a protective assessment (not more than 5% potentially affected species) of the risks associated with dimethoate when applying the SSD concept (Posthuma et al. 2002). The small difference between the HD5 lower 95% confidence limit and 48 h LD50/10 may be alleviated by slightly increasing the bridging factor. Furthermore, a relationship of bee body weight and acute toxicity was established for dimethoate. Such regression methods could be applied, using bee traits such as weight, to extrapolate toxicity from surrogate species (Figure 19).

Table 29: Acute contact toxicity of the dimethoate formulation Perfekthion® towards several European bee species.

Species	LD50 (µg a.i./bee)	95% CI (µg a.i./bee)	Mean fresh weight (mg)	LD50 fresh weight-normalised (µg a.i./g bee)	95% CI fresh weight-normalised (µg a.i./g bee)	Mean dry weight (mg)	LD50 dry weight-normalised (µg a.i./g bee)	95% CI dry weight-normalised (µg a.i./g bee)
L. politum	0.02	0.01-0.03	3.6	6.35	3.45-9.25	NA	NA	NA
O. cornifrons	0.09	0.02-0.20	NA	NA	NA	22.2	4.05	0.90-9.01
L. malachurum	0.12	0.09-0.14	11.0	9.80	7.84-11.76	NA	NA	NA
A. mellifera	0.18	NA	99.7	1.82	NA	17.0	10.66	NA
C. hederae ♂	0.23	0.22-0.25	42.9	5.45	5.02-5.88	NA	NA	NA
B. lapidarius	0.31	0.03-0.59	143.5	2.16	0.22-4.11	NA	NA	NA
A. gallica	0.68	0.39-0.97	126.2	5.36	3.07-7.65	NA	NA	NA
A. flavipes	0.73	0.07-1.39	47.3	15.44	1.57-29.31	21.6	33.78	3.44-64.11
C. hederae ♀	1.14	0.72-1.57	105.5	10.84	6.83-14.85	43.4	26.35	16.61-36.09
O. lignaria	1.21	1.05-1.57	92.3	13.11	11.38-17.01	29.4	41.16	35.71-53.40
O. bicornis ♂	1.71	1.37-2.04	37.7	45.27	36.31-54.22	17.6	96.90	77.73-116.07
O. bicornis ♀	4.29	3.72-4.91	93.6	45.89	39.80-52.47	30.4	141.46	122.68-161.73
B. terrestris	5.13	4.10-6.15	205.0	25.00	20.00-30.00	55.8	91.87	73.49-110.2

Table adapted from Uhl et al. (2016). Species were sorted LD50 value (µg a.i./bee) in ascending order. Where multiple LD50 values were available they were summarised using the geometric mean. Fresh and dry weight-normalised values were calculated by dividing the LD50 by the respective weight. LD50 values for *A. mellifera*, *O. cornifrons* and *O. lignaria* were extracted from peer-reviewed literature. LD50 values for *A. gallica*, *B. lapidarius*, *C. hederae* ♂ and *L. politum* are unpublished results from the same working group that were generated after the publication of the above-mentioned article using an identical study design.

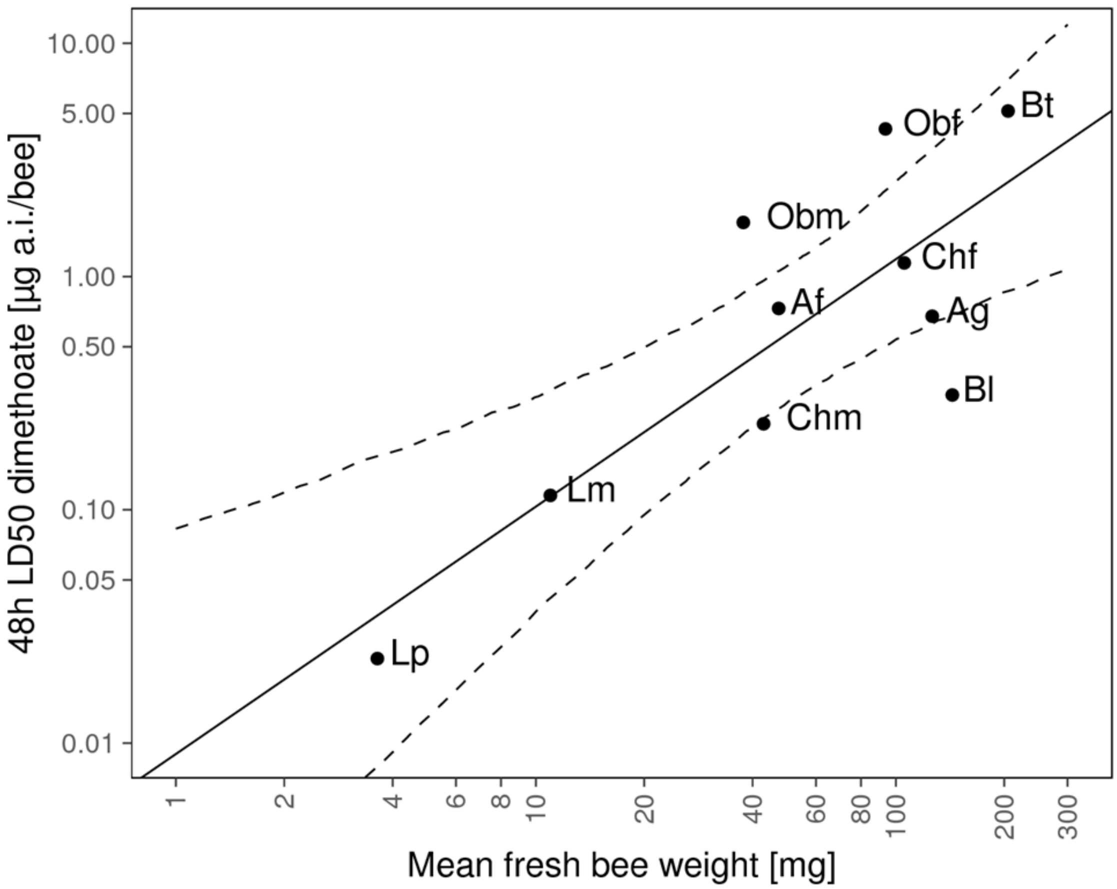
Figure 18: Species sensitivity distribution of dimethoate calculated from multiple bee species' acute sensitivity.



Footnote: ● & ○ denote 48 h LD50 values of bee species (○ are literature values). Species names are aligned by sensitivity in ascending order from bottom to top on the same y-axis coordinate as their respective ●/○. Dashed lines enclose parametric bootstrap 95% CI (1000 iterations). Blue, transparent lines display all parametric bootstrap samples. ◆ marks the HD5 value, ▲ the lower limit HD5. The proposed regulatory threshold of honey bee LD50/10 is indicated by the dotted line.

Source: own illustration. Adapted from Uhl et al. (2016) and complemented with additional data

Figure 19: Relationship between fresh bee weight and sensitivity towards dimethoate.



Footnote: Dots mark weight and sensitivity of the following species: Af - *Andrena flavipes*, Ag – *Andrena gallica*, Chm - *Colletes hederae* ♂, Chf - *Colletes hederae* ♀, Lm - *Lasioglossum* malachurum, Lp – *Lasioglossum politum*, Obm - *Osmia bicornis* ♂ , Obf - *Osmia bicornis* ♀ , Bl – *Bombus lapidarius*, Bt - *Bombus terrestris*. Both axes on logarithmic scale. Dashed lines enclose parametric bootstrap 95% CI (1000 iterations).

Source: own illustration. Adapted from Uhl et al. (2016) and complemented with additional data

However, Uhl et al. (2016) also noted that relative susceptibility varies for different pesticides and that the bridging factor might need to be adapted for other pesticides or pesticide groups after sufficient data is collected. Especially for neonicotinoids with a high honey bee toxicity these data are still missing. With the current data it is therefore difficult to extrapolate acute toxicity data of a specific pesticide from the honey bee to a specific wild bee species (Helson et al. 1994; Biddinger et al. 2013). Consequently, there is not only a need to screen more wild bee species for their sensitivity towards one standard substance (e.g. dimethoate) but also to test single wild bee species with an array of pesticides. Arena & Sgolastra (2014) could only gather limited LD50 values for the European wild bee species *Bombus terrestris* (10 pesticides, 24 h LD50) and 16 *Megachile rotundata* (16 pesticides, 48 h LD50). In a recent study Heard et al. (2017) established dose-response relationships of *A. mellifera*, *B. terrestris* and *O. bicornis* for clothianidin, dimethoate, tau-fluvalinate, cadmium and arsenic (48, 96 and 240 h) after oral application. They found a less than 2-fold difference in LC50 values between honey bee and both wild bee species and concluded that the honey bee may be a sufficient proxy for wild bees if an adequate bridging factor is applied, although no suggestion was made. However, they also stated that there might be chemicals that are exceptions and also that delayed effects after continued exposure should be recognised in risk assessment. In another acute laboratory study sixteen insecticide formulations most of which are applied in common crops in Germany (and to-be-registered flupyradifurone) were tested with *Osmia bicornis* (Uhl et al. in prep.). When comparing contact toxicity of these compounds to the wild bee species with existing regulatory honey bee toxicity endpoints large pesticide-specific differences became evident: The sensitivity ratio of 48 h $LD50_{A.\ mellifera}/LD50_{O.\ bicornis}$ varied from 0.02 to around 18, spanning nearly three orders of magnitude (Table 30). For 13 out of 15 evaluable insecticides a bridging factor of 10 would have been enough to extrapolate from honey bee sensitivity as proposed by Arena & Sgolastra (2014). However, the suggested procedure might not be protective for all products and also for all wild bee species, since only a limited number of species was tested so far and data are lacking especially for small species. Furthermore, honey bee endpoints for active ingredients and formulations differed considerably for some substances (Table 30; e.g. flupyradifurone, thiacloprid).

Table 30: Comparison of acute, contact toxicity of several insecticides used in major German crops towards *Apis mellifera* and *Osmia bicornis.*

Pesticide	Product	LD50 formulation		Sensitivity ratio of formulation	LD50 a.i.
		Osmia bicornis µg a.i./bee	*Apis mellifera*		*Apis mellifera* µg a.i./bee
zeta-cypermethrin	Fury® 10 EW	0.132	0.002	0.02	NA
spinosad	SpinTor®	2.059	0.050	0.02	0.004
indoxacarb	AVAUNT® 150 EC	1.264	0.080	0.06	0.068
dimethoate	PERFEKTHION®	1.319	0.111	0.08	0.100
pirimicarb	Pirimor®	115.067	36.100	0.31	NA
alpha-cypermethrin	FASTAC® SC	0.244	0.090	0.37	0.030
lambda-cyhalothrin	Karate® Zeon	0.136	0.055	0.41	0.038
deltamethrin	Decis® Forte	0.057	0.029	0.51	0.002
chlorpyrifos	Pyrinex®	4.188	3.190	0.76	0.068
beta-cyfluthrin	Bulldock®	0.035	0.032	0.90	0.012
flupyradifurone	Sivanto® SL 200 G	10.586	17.100	1.62	>200.000
acetamiprid	Mospilan® SG	1.719	9.260	5.39	8.090
imidacloprid	Confidor® WG 70	0.031	0.245	7.83	0.081
chlorantraniliprole	Coragen®	5.918	>100.000	16.90	>4.000
thiacloprid	Calypso®	1.159	20.813	17.96	38.820
etofenprox	Trebon® 30 EC	0.177	NA	NA	0.015

Table adapted from Uhl et al. (in prep.). Sensitivity ratio for the formulated products was calculated as LD50 *A. mellifera*/LD50 *O. bicornis*. Pesticides sorted by sensitivity ratio. Honey bee data derived from regulatory documents (e.g. EFSA conclusions, EC rapporteur member state draft/renewal assessment reports) or personal communications from UBA. Flupyradifurone is in the progress of authorisation in several EU member states.

During the conduct of acute contact bee studies with active chemicals wetting agents are used for the application of the droplets to the bees' thorax. Since their effectiveness differs they may influence toxicity even if not causing toxic effects by themselves. The use of wetting agents in bee acute testing should also be regulated more clearly. When comparing *Osmia bicornis* toxicity of dimethoate using five different wetting agents (Triton X-100, Tween 80, Etalfix Pro, Acetone), Eschenbach (2016) found variation in 48 h LD50s of up to a factor of 6. Thompson (2016) further argued that for interspecific sensitivity comparisons of bee species acute toxicity endpoints should be calculated on a per weight basis to exclude the effect of body mass on toxic responses (also see Table 29 for a comparison of LD50 values to fresh or dry weight-normalised LD50 values). In a re-analysis of the dataset of Arena &

Sgolastra (2014) they showed a reduction of 95th percentile of sensitivity ratio from 10.7 to 5.0 (contact and oral exposure) when normalising LD50 by weight. It has to be noted that while this approach is scientifically sound interspecific body size differences should then be accounted for in exposure estimation (see 5.4; Detailed description of exposure estimation) instead of first tier effect assessment.

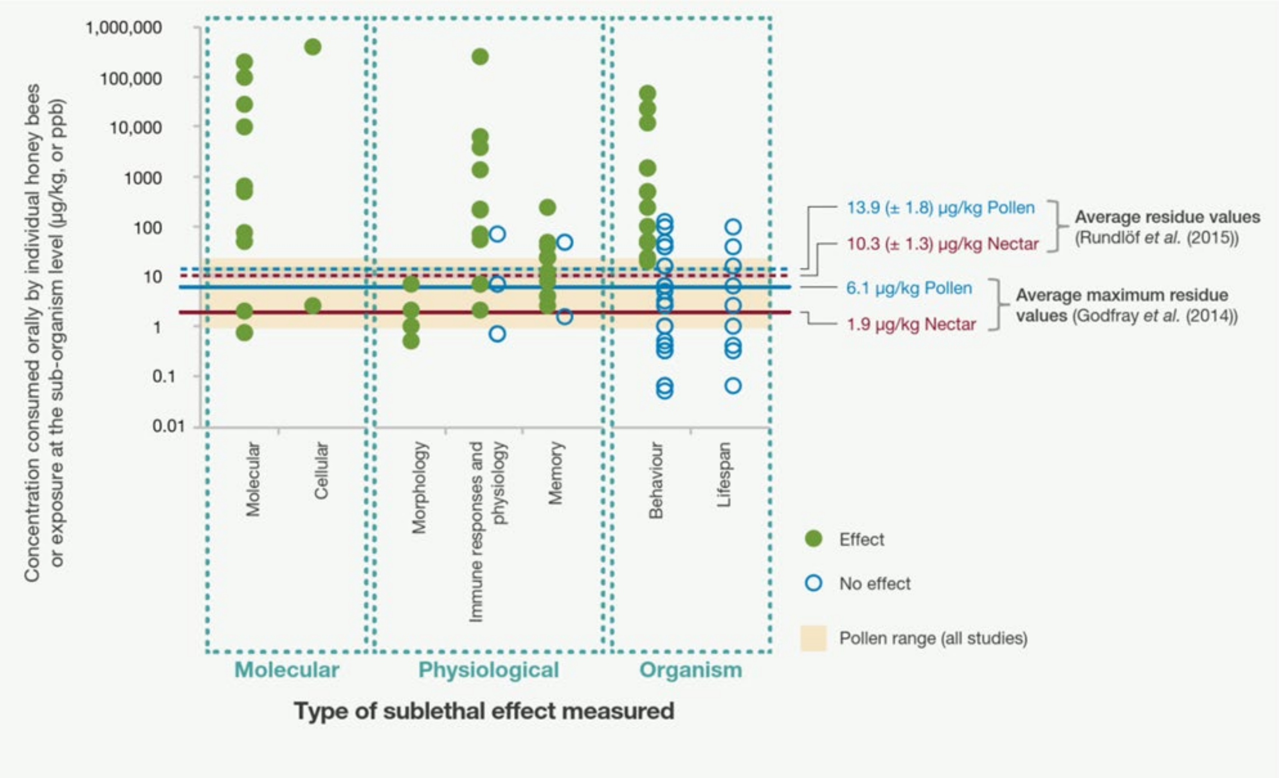
Mixture toxicity has also been investigated in laboratory studies. Sgolastra et al. (2016) tested *A. mellifera*, *B. terrestris* and *O. bicornis* for the combined effect of the neonicotinoid clothianidin and the ergosterol-biosynthesis-inhibiting fungicide propiconazole: They found synergistic mortality effects in all three species after oral exposure. Robinson et al. (2017) tested the same species with multiple combinations of clothianidin, dimethoate, propiconazole, tau-fluvalinate, cadmium and arsenic for up to 240 h. They could detect synergistic mortality of some mixtures but also found antagonistic effects for all three species (e.g. clothianidin and dimethoate). Furthermore, Tsvetkov et al. (2017) found increased mortality after 24 h of oral exposure to a mixture of clothianidin or thiamethoxam with the fungicide boscalid compared to treatments with only clothianidin or thiamethoxam.

6.1.1.2 Sublethal effects

There have been numerous studies investigating toxic effects of pesticides below lethal doses on honey bees. Since the goal of this report is to evaluate pesticide effects preferentially on wild bee species, these studies will not be discussed here. Please see the following review articles for further information on sublethal honey bee effects: Blacquiere et al. (2012), Godfray et al. (2014, 2015), Goulson (2013a), Pisa et al. (2015), Wood & Goulson (2017).

Considering field-relevant doses of pesticides (see 5.2 Evidence of exposure by residue levels of pesticides) studies that demonstrated effects at unreasonable high doses were not included in this section or are indicated as such. Figure 20 demonstrates that bee effect studies should preferably investigate field-realistic doses. Furthermore, it shows the wide range of sublethal neonicotinoid effects that were detected in individual honey bees.

Figure 20: Overview of reported neonicotinoid pesticide exposure and effect doses of individual adult honey bees.



Footnote: Information of following documents is summarised: EFSA (2013), Godfray et al. (2014), Rundlöf et al. (2015), USEPA (2014).

Source: IPBES (2016)

There have been several studies of sublethal effects with bumble bees. One ecologically relevant parameter to measure is the reproductive capability of a colony. Baron et al. (2017) collected queens of four species (*B. terrestris*, *B. lucorum*, *B. pratorum* and *B. pascuorum*) and fed them to syrup treated with field realistic concentrations of thiamethoxam (1 and 4 ng/mL) for 14 days. The higher concentration reduced feeding in two out of four species. Furthermore, queens in this treatment had smaller oocytes than in control and low treatment which may impair their reproductive success. In a laboratory feeding study Tasei et al. (2000) exposed *B. terrestris* micro-colonies through syrup and pollen for 85 days to imidacloprid doses of 10 and 6 ng/g respectively and 25 and 16 ng/g respectively in two treatments which would be both worst-case field exposures. Both treatments substantially decreased worker survival. The lower treatment caused a reduction in brood production whereas both treatments resulted in decreased larval ejection by workers. In another chronic laboratory study queenright *B. terrestris* colonies were fed 10 ng/mL imidacloprid in sugar water for 42 days which caused the eclosion rate of new worker to drop to zero after 21 days while death rate increased leading to colony collapse (Bryden et al. 2013). Laycock et al. (2012) fed *B. terrestris* micro-colonies with sugar solutions containing 0 to 125 ng/mL imidacloprid. They found several dose-dependent effects: After 13 days a dose as low as 1.27 ng/mL caused a decline of layed eggs by 42%. Furthermore, consumption rates of pollen and sugar solution decreased. In a similar experiment Laycock & Cresswell (2013) used queenright *B. terrestris* colonies to investigate effects of pulsed imdacloprid exposure (0-125 ng/mL). After 14 days of exposure they also found reductions in brood production at doses between 0.38 and 12.7 ng/mL as well as pollen and sugar water feeding reduction. However, there was partial recovery of the brood reduction 14 days after the exposure period. Elston et al. (2013) used a setup where they treated both a pollen paste and sugar solution with 1 and 10 ng/g thiamethoxam and exposed bumble bees for 28 days. Both treatments reduced sugar solution consumption. There was a decrease in nest-building activity as well as egg and larvae production in the 10 ng/g treatment. Larvae production was actually completely stopped. Scholer & Krischik (2014) studied *Bombus impatiens* colonies in a greenhouse which were exposed to imidacloprid- and clothianidin-treated sugar syrup for 11 weeks. Queen mortality increased after six weeks from 0% in the control to 37.5% for both pesticides at 50 ng/g and from 12.5% and 25% to 50% and 62.5% for imidacloprid and clothianidin, respectively, at 100 ng/g. It has to be noted that these exposure levels are unrealistically high. Furthermore, after 11 weeks increased mortality at doses as low as 20 ng/g was observed whereas 100 ng/g lead to 100% mortality for both pesticides. Colonies in treatments above 10 ng/g imidacloprid and 20 ng/g of clothianidin gained significantly less weight.

Pesticide effects on foraging activity have also been detected in bumble bees, and many recent studies focused on the effects of neonicotinoids. Morandin & Winston (2003) exposed *B. impatiens* colonies to imidacloprid-treated pollen mixed with sugar solution. This lead to prolonged visitation time of 30 flowers in an artificial flower array at a worst-case scenario dose (30 ng/g pollen). Furthermore, cognitive abilities of bumble bees can be affected by pesticides. In an acute exposure scenario *B. terrestris* workers showed effects only at worst-case levels of thiamethoxam-treated sugar solution. However, after 24 days of exposure even a field-realistic dose of 2.4 ng/g lead to short-term memory impairment and decreased learning speed in proboscis extension reflex (PER) assays: Thiamethoxam-exposed bees learned the task slower and their first response happened later than in control bees (Stanley et al. 2015b). In a multi stressor study Piiroinen et al. (2016) studied the effects of clothianidin in combination with *Nosema ceranae* (microsporidian parasite) inoculations on *B. terrestris* micro-colonies. Chronic exposure to 1 ng/g clothianidin sugar solution did not result in sublethal effects on the measured parameters (e.g. fecundity, food collection, learning). Furthermore, bees did not become infested with the parasite. Similar results were found by Piiroinen & Goulson (2016) in an experiment were bumble bees were exposed to a sugar solution containing 4 ng/g clothianidin: There was no substantial impairment of olfactory learning in contrast to honey bees' reaction and nearly no infestations with *N. ceranae*. Interactions of different types of stressors are relevant since they might act on wild bees synergistically, i.e. cause adverse effects that are more severe than the added effects of the individual stressor. In such a combination study the impact of thiamethoxam and clothianidin and the parasite *Crithidia bombi* on hibernation of *B. terrestris* queens was investigated (Fauser et al. 2017). Pesticide treatment of both 4 ng/g thiamethoxam and 1.5 ng/g clothianidin was delivered through spiked sugar water and pollen patties. The parasite as well as both neonicotinoids reduced hibernation success during a four months hibernation period. However, parasite and pesticide effects were not additive but rather asynchronous with the pesticides overriding the parasite effect. Moreover, neonicotinoid treatment increased hibernation weight loss (Fauser et al. 2017). The same treatment combination resulted in synergistic reduction in queen survival after nine weeks of exposure in a colony-level study. Neonicotinoid treatment alone lead to colony development effects such as decreased reproductive success, shortened worker lifespan and truncated worker production (Fauser-Misslin et al. 2014).

Aside from bumble bees, sublethal laboratory pesticide effects have also been studied in a few experiments with other wild bee species. Abbott et al. (2008) treated pollen provision of *O. lignaria* larvae with imidacloprid and monitored overwintering until emergence. They found prolonged development times in an imidacloprid treatment that represents a worst-case scenario (30 ng/g). Larvae of *Megachile rotundata* that were exposed to clothianidin in a similar setup showed no sublethal effects. Furthermore, the effect of the neonicotinoids clothianidin and thiamethoxam on reproductive parameters was investigated in *O. bicornis* (Sandrock et al. 2014). Treatment bees were chronically and orally exposed to a nectar substitute spiked with both pesticides in field-relevant doses for their entire active flight period in an environmental chamber where they could forage, mate and lay eggs freely. Exposed females completed fewer brood cells whereas larval mortality increased, leading to a decrease in hatched offspring after hibernation. Since larval food provisions, which consist mostly of pollen, were not spiked with a pesticide, this implies an effect on the reproductive performance of females. Furthermore, the treatment offspring's sex ratio was biased towards males in contrast to the control. In consequence there was a 2.3-fold reduction in hatched females. However, in a study where *O. bicornis* larvae pollen provisions were injected with 1, 3 and 10 ng/g clothianidin there were no effects on either developmental time, overwintering survival or hatched adult weight (Nicholls et al. 2017). The authors hypothesised that neonicotinoid effects on populations of this species might be mediated by reproductive effects on adults as shown by other studies (e.g. Sandrock et al. 2014, Rundlöf et al. 2015).

Perspective: Study designs for FVI sublethal effect testing

Test conditions

Sublethal effects should be assessed in chronic toxicity tests since worst-case lethal effects are already investigated in acute toxicity tests in first tier pesticide risk assessment. Test duration should be adapted according to the ecotoxicological endpoint. In the above-mentioned scientific studies test duration was quite variable. Nicholls et al. (2017) tested *O. bicornis*' larval development, overwintering and hatching in a study that lasted over 300 days whereas Baron et al. (2017) used a four week setup (two weeks exposure) to determine effects on bumble bee oocyte development. A test should be long enough to allow for detection of an effect on this specific ecological parameter. Furthermore, other test conditions such as temperature or artificial day length should be chosen with respect to the test species' ecology. Mode of application should be oral since this is more realistic than contact exposure in chronic scenarios.

Test species

Aside from the honey bee, *B. terrestris* and *O. bicornis* have been established as test organims for sublethal toxicity tests. They are representative of social and solitary bee species, respectively. However, it might be necessary to consider additional test species in cases where effects on these two species cannot generally be extrapolated to all ecologically similar groups of bees (i.e. vulnerable groups). It is more complicated to refer potential test species for other FVI taxa. For butterflies and moths there are several species which have been used as test organims (usually in lethal effect studies) that might be adequate for sublethal effect studies: *Aglais urticae*, *Aglais io*, *Pieris brassicae* or *Pieris rapae* (butterflies) and *Helicoverpa zea*, *Mamestra brassicae*, *Plutella xylostella, Cydia pomonella* (moths). However, most of these species are agricultural pests and may not be representative for non-target species for the evaluation of ecologically meaningful sublethal effects.

Endpoints

There are numerous ecological parameters that might be chosen as ecotoxicological endpoints. The above-mentioned studies investigated effects on e.g. proboscis extension reflex, feeding, flower visitation, oocyte development, oviposition, nest-building activity, colony weight gain, larval development, overwintering performance, hatching rate or worker life span. As these effects may often be correlated it is important to choose a minimal number of ecologically meaningful endpoints which reflect changes that would lead to population declines (i.e. generational effects).

Test protocols

EFSA (2013) proposed test protocols to assess the chronic oral toxicity towards honey bee adults and larvae as well as accumulative oral toxicity. They further made suggestions for a chronic bumble bee micro-colony test and a solitary bee chronic oral toxicity test. These test protocols could be adapted to incorporate (more) sublethal effect assessments. The above-mentioned studies may also be used as templates. Test designs should be adequate to detect generational effects with high statistical power.

Uncertainties

It is unclear to which extent sublethal laboratory effects can be extrapolated to the field. Furthermore, the representativeness of possible test species for their own or even other FVI groups has not been sufficiently studied. As there is not enough information availabe to address these issues at the moment, uncertainties should be reflected in the use of a conservative safety factor.

6.1.2 Field & semi-field studies

6.1.2.1 Introduction

In 2013 the European Commission restricted the neonicotinoid compounds imidacloprid, clothianidin and thiamethoxam in use because of high acute risks for bees. Since then several complex field studies have been carried out to further the understanding of neonicotinoid effects on bees, honey bee and wild bee species, in the agricultural landscape. Therefore, the overwhelming majority of bee field studies were conducted with this pesticide class. Most research was performed with the honey bee. Honey bee field effects will not be elaborated on in particular in this report: Several colony-level honey bee studies found limited to negligible effects (Pilling et al. 2013; Cutler et al. 2014; Dively et al. 2015; Rundlöf et al. 2015). Wood & Goulson (2017) argue that honey bee effects from those studies are hardly translatable to wild bee species (i.e. all other European bee species) because of the substantial differences in social structure and sheer numbers in a population. Stoner (2016) also concluded that ecological differences between honey bee and bumble bees are significant and that research should focus on these critical differences to adequately assess pesticide risks towards bumble bees. Please see the following review article for further information on honey bee field effects: Blacquiere et al. (2012), Godfray et al. (2014, 2015), Goulson (2013a), Pisa et al. (2015).

The following studies established the impact of pesticides on different parameters that are crucial for the maintenance of stable wild bee populations.

6.1.2.2 Reproduction

There have been several recent field studies that investigated wild bee reproduction and colony growth effects when exposed to mass-flowering crops. Cutler & Scott-Dupree (2014) set up *Bombus impatiens* colonies next to maize fields (n = 4). Colonies were exposed at maize fields for 5-6 days and then transported to an area of semi-natural habitat for 30-35 days. Treatment colonies produced fewer workers than those set up at organic farms. Bumble bees collected less than 1% of pollen from maize and neonicotinoid pollen residues were very low (0.4 ng/g).

Rundlöf et al (2015) conducted a large-scale field experiment to study the effects of clothianidin-treated oilseed rape on two wild bee species. They selected 16 fields across Sweden and paired them by similar landscape composition. In each pair one field was seed-treated with clothianidin (10 g/kg) and one was sown without treatment. Twenty-seven *Osmia bicornis* cocoons in three nesting aids (12 females, 15 males) as well as six *Bombus terrestris* colonies were placed at each field before oilseed rape bloom. There were no brood cells constructed by *Osmia bicornis* next to treated fields whereas *Osmia bicornis* adjacent to untreated fields showed brood cell building. Bumble bees next to treated fields experienced reduced colony growth and reproductive output. Numbers of queens and workers/males were significantly reduced.

Sterk et al. (2016) performed a similar study: In Northern Germany they selected two areas of 65 km^2 where the only flowering crop was winter oilseed rape. Seed treatments of 10 g/kg clothianidin (as used by Rundlöf et al.) were applied in one area while the other area was left untreated. *Bombus terrestris* colonies were placed at six sites within each area. Colonies were left to forage from April to June, through bloom. They found no differences in colony weight growth, number of workers produced or reproductive output as measured by the production of new queens. Within the same setup Peters et al. (2016) placed 1500 *Osmia bicornis* cocoons at each site. They also found no differences between treated and untreated areas in nest building activity and even positive effects on other reproductions parameters such as emergence after overwintering. However, as noted by Wood & Goulson (2017) among some major differences between the Swedish and the German study is that Rundlöf et al. (2015) used spring-sown oilseed rape while Sterk et al. (2016)/Peters (2016) et al. used winter-sown oilseed rape. In the German study there was substantially more time for clothianidin to dissipate into the soil before being taken up by the oilseed rape plants leading to lower exposure of nectar and pollen. Moreover, the German study features no true replication at area level whereas the Swedish study incorporates substantially smaller numbers of individuals and features no nested sites within field areas.

In a study by Ellis et al. (2017) queenright *B. terrestris* colonies were placed next to raspberry farms. Treatment raspberry farms were applied with thiacloprid by spray (Calypso 480 g a.i./L, application volume up to 250 mL/ha) whereas control farms were not applied with pesticides. Colonies were left at the farm sites to forage for two weeks and then moved to flower-rich heather moorland sites. Thiacloprid doses were determined as up to 771 ng/g in pollen and 561 ng/g in nectar in a separate analytical sampling run at some of the farming sites a year later which complicates direct linking of exposure and effects. Treated colonies were more likely to die prematurely. Furthermore, treated colonies at flower-rich sites had a lower final weight and produced less offspring than control colonies.

Moffat et al. (2015) tested *B. terrestris* colonies in a field setting where they could forage freely in a landscape with low pesticide input but exposed them through spiked sugar solution to 2.1 ng/g imidacloprid for 43 or 48 days. This reduced the number of surviving bees and lead to a reduction of colony growth and brood. Using a similar setup Moffat et al. (2016) investigated the impact of field-relevant levels (2.5 ng/g) of imidacloprid, thiamethoxam and clothianidin on bumble bee colonies. Thiamethoxam reduced the number of live bees and shifted the sex ratio towards males. Clothianidin actually increased queen production whereas imidacloprid and thiamethoxam reduced the overall number of brood cells after 35 days. Whitehorn et al. (2012) exposed *Bombus terrestris* colonies to imidacloprid for two weeks in the laboratory by spiking pollen (6 and 12 ng/g) and nectar (0.7 and 1.4 ng/g) with imidacloprid. Afterwards, colonies were moved outside to forage for six weeks. Imidacloprid-treated bumble bees grew more slowly and the queen production was reduced by 85% in these colonies. Larson et al. (2013) enclosed *B. impatiens* to forage on clothianidin- and chlorantraniliprole-treated lawn for 6 days. Pesticide applications were performed according to the label rate which resulted in doses of 171 ± 44 ng/g clothianidin in nectar of white clover (*Trifolium repens*) which grew on the lawn (no information for chlorantraniliprole). Afterwards, colonies were moved to a safe foraging site where no insecticides were applied and bees could forage freely for 6 weeks. Clothianidin treatment stopped queen production and caused a significant reduction of colony weight whereas chlorantraniliprole treatment resulted in no adverse effects and even increased the number of adult workers. In a similar experiment Gels et al. (2002) investigated the effects of imidacloprid granular and spray label rate applications on *B. impatiens*. They found no side-effects of granule and spray treatments when there was irrigation after application. However, non-irrigated spray residues caused a reduction of workers, honey pots and brood chambers after 30 days of exposure.

6.1.2.3 Foraging

Gill et al. (2012) fed imidacloprid-treated sugar solution (10 ng/g) to *Bombus terrestris* colonies for four weeks. Colonies were kept indoors but bumble bees could leave and forage outdoors through access tubes. Imidacloprid treatment increased the number of foraging trips but workers collected less pollen and the time for successful trips increased. Furthermore, the proportion of successful trips was reduced from 82% in the control to 59% in the treatment. Consequently, imidacloprid-treated colonies collected fewer pollen and subsequently grew less. Gill & Raine (2014) used an identical setup to Gill et al. (2012) in terms of exposure route, dose and time as well as the ability of bumble bee to forage freely outside. As in their previous study, treatment workers went on more foraging trips. Foraging efficiency decreased throughout the study.

Feltham et al. (2014) treated sugar solution and pollen with low doses of imidacloprid (0.7 ng/g and 6 ng/g, respectively). *Bombus terrestris* colonies were exposed for two weeks. Afterwards, colonies were set up in an urban area in Scotland and foraging activity was monitored for four weeks. Despite no substantial difference in foraging bout length between treatment and control, treatment workers collected 31% less pollen per hour. Furthermore, treatment resulted in reduced successful foraging trips (41% compared to 65% in control)

Stanley et al. (2015a) exposed *Bombus terrestris* to 2.4 or 10 ng/g thiamethoxam in sugar solution for 13 days. Then, colonies were moved to exclusion cages where bumble bees could forage on apple flowers. Bumble bees exposed to 10 ng/g showed increased time of foraging trips, visited less flowers and had a lower frequency of successful trips. Stanley & Raine (2016) fed *Bombus terrestris* 10 ng/g thiamethoxam sugar solution for nine to ten days. Thereafter, colonies were moved to a flight arena where bumble bees were offered two common bird's-foot trefoil *Lotus corniculatus* and one white clover *Trifolium repens* plants. Workers were individually released and their flower interactions recorded. While more treatment workers showed increased foraging behaviour, control workers needed less visits since they handled flowers more effectively. Stanley et al. (2016) investigated homing ability of *B. terrestris* after chronic exposure to 2.4 ng/g thiamethoxam in sugar water (5-43 days). In pesticide treated bees the homing rate was actually higher than in the control (92% and 67% respectively).

Arce et al. (2016) set up *Bombus terrestris* colonies in a parkland area and fed them clothianidin-treated sugar solution (5 ng/g). They provided no pollen and only half the volume of sugar solution they estimated the colonies would need for the experimental duration so that worker would be forced to forage. Their results show only minor changes in foraging activity and pollen collection. Furthermore, colony weight gain or brood number was not affected by the end of the experiment but numbers of workers, drones and gynes were reduced in the treatment.

6.1.2.4 Immune system

There have been several studies linking neonicotinoid exposure to increased disease and parasite susceptibility in honey bees (e.g. Alburaki et al. 2015, Dively et al. 2015, Pettis et al. 2012, Vidau et al. 2011). Such effects were to this day not studied in wild bees in field scenarios (for laboratory immune effects see section 6.1.1.2 Sublethal effects). Since they have a very similar nervous and immune system to honey bees there is a distinct possibility that neonicotinoids make wild bees also more prone to disease and parasites (Wood & Goulson 2017). Furthermore, fungicide effects on immune functions should not be neglected: 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.

6.1.3 Source-sink effects

As mobile species with flying adult stages FVIs can easily move between in-field and off-field habitats. However there are not many studies available for a detailed analysis for FVIs (e.g. bumble bees; Osborne et al. 1999) but more for ground-dwelling beetles (e.g. carabid beetles; Holland et al. 2004). This spatial factor has to be considered when investigating pesticide effects of FVI populations. Migration from semi-natural off-field habitats to pesticide-treated in-field areas can result in source-sink dynamics: Individuals from a sustaining habitat migrating to a non-sustaining habitat consequently subsidising the sink population but also possibly depleting 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 that landscape-scale effects of pesticides cannot be sufficiently estimated using small-scale data. Topping et al. (2014) simulated beetle and spider population responses to pesticide applications and concluded that small-scale plot experiments are severely underestimating landscape-level effects. In another modelling study Topping et al. (2015) evaluated pesticide impacts on a beetle species and found that source-sink dynamics resulted in off-field effects even when no pesticides were present in these habitats but only in crop fields. Furthermore, they showed the impact increasing over multiple years. Migratory population dynamics in time and space are difficult to detect using field experiments due to limited run time and restricted spatial scale. Therefore, landscape-scale modelling approaches represent promising methods to assess source-sink effects of pesticides. Please see section 7.2 for a feasibility study on agent-based modelling of FVI species in the European landscape.

6.1.4 Adjuvants

Commercial pesticide formulations consist of an active ingredient and several additional substances. These so-called inert ingredients, co-formulants or adjuvants are deployed to optimise the efficacy of the pesticide product. However, they can actually enhance the toxicity of the active ingredient towards non-target species or cause toxic effects by themselves (Mullin et al. 2015). Zhu et al. (2014) assessed the oral toxicity of the solvent N-methyl-2-pyrrolidone towards honey bee larvae and found 100% mortality within 24 h caused by a 1% solution. Furthermore, adjuvants can also impair physiological responses in bees at below lethal doses as shown by Ciarlo et al. (2012): Oral exposure of worker honey bees to 1% solutions of several organosilicone and non-ionic surfactants resulted in reduced learning ability in proboscis extension reflex (PER) assays. Similar to active ingredients, adjuvants have been shown to weaken the immune system of bees. In a chronic honey bee brood study the organosilicone compound Sylgard 209 (10 nL/mL in bee diet) caused only slightly higher mortality than the control. However, in combination with viral agents mortality increased synergistically compared to the viral agents themselves (Fine et al. 2017). The few available studies on adjuvant toxicity towards FVIs indicate that these substances have the potential to induce a similar range of effects as the actual active ingredient.

In the EU pesticide registration process manufacturers need to provide toxicity information for the active ingredient and the respective formulated product. However, they are not required to disclose details on adjuvant composition in a formulation. Therefore, toxicity assessment of these additives is not possible. This is problematic for FVI risk assessment as for certain formulations the inert ingredients might be more toxic than the active ingredient (Mullin et al. 2015).

6.1.5 Indirect effects

Aside from direct effects pesticides can also impact bees indirectly through trophic interactions. Habitat quality may be reduced by reduction of food and nesting resources. Müller et al. (2006) hypothesised that one of the main factors causing bee declines is the decrease in diversity and quantity of flower resources caused by habitat destruction and agricultural land use practices. Scheper et al. (2014) determined host plant preference of wild bee species using pollen loads collected by specimens that are stored in entomological museum collections. They combined pollen load data collected before the onset of bee declines with population trends of wild bees and identified pollen sources. Decline of preferred host plant species was identified as one of two main factors associated with bee species declines. In a field experiment simulating pesticide input into field margins, flower density of the common buttercup *Ranunculus acris* which is host to the oligolectic bee species *Osmia florisomnis* and visited by more than 100 insect species was reduced by 85% by a sulfonylurea herbicide (Weiner et al. 2011; Schmitz et al. 2013). Furthermore, the herbicide application lead to shifts in plant community composition and supressed flower formation in meadow pea *Lathyrus pratensis* and bush vetch *Vicia sepium* (Schmitz et al. 2014a, b). The herbicide directly affected frequency of the hemiparasitic rattle *Rhinanthus alectorolophus* leading to total population decline. Such an effect may impact many bumble bee species who use *Rhinanthus* species as a central food source and other FVIs since *Rhinanthus* species are keystones of meadow food webs (Kwak 1980; Hartley et al. 2015). In another study Bohnenblust et al. (2015) found that dicamba application of 1% of the field rate delayed flowering onset and reduced number of flowers in alfalfa *Medicago sativa* and boneset *Eupatorium perfoliatum*. Furthermore, plants that did flower were visited less often by FVIs, mostly honey bees but also lepidopteran, fly, wild bee and beetle species.

Indirect effects may even be more subtle. Bartlewicz et al. (2016) studied effects of multiple fungicides on nectar microorganisms of common toadflax *Linaria vulgaris*. This plant often grows alongside arable fields and is visited by bumble bee species. Prothioconazole and tebuconazole substantially inhibited the growth of three nectar yeasts at concentrations between 0.06 and 0.5 µg/mL which consequently may reduce flower attractiveness to bee species. However, more research is needed to evaluate this hypothesis.

6.1.6 Ecosystem services (Pollination/biodiversity)

There is little research linking pesticide effects on bees and pollination services. Stanley et al. (2015a) exposed *Bombus terrestris* to realistic doses of thiamethoxam and allowed them to forage on apple trees in a semi-field cage experiment. Apple seed production was subsequently reduced by 36% showing first evidence of a direct pesticide pollination effect in a field setting. Switzer & Combes (2016) investigated the effect of acute imidacloprid poisoning on sonication (i.e. buzz pollination) performance in *Bombus impatiens* on tomato *Solanum lysopersicum*. At doses of 0.515 or 5.15 ng in 10 µL of sugar solution bumble bees rarely performed foraging behaviour anymore.

It is difficult to directly detect bee population effects in field experiments since it would take years and a massive sampling campaign to collect the necessary data. However, in a meta-analysis Woodcock et al. (2016) related distribution monitoring data of 62 bee species in the UK over an 18 year period to neonicotinoid use in oilseed rape. They could show that population persistence was negatively affected in bee species that forage on oilseed and those that usually do not while the effect was three times stronger in oilseed rape foragers. Therefore, they linked neonicotinoid use in a mass-flowering crop to bee species decline to a certain degree.

Perspective: Study designs for FVI field & semi-field effect testing

Test conditions

Realistic test conditions should be ensured by choice of test crop and landscape surroundings. Test dates should be chosen with regard to the test organism's phase of highest flower visitation activity or active phases of sensitive life stages (e.g. herbivore life stages of lepidopterans). Test duration should be adapted according to the ecotoxicological endpoint. A test should be long enough to allow for the detection of an effect on this specific ecological parameter. Pesticide applications should be performed according to the manufacturer’s instructions.

Test species

Aside from the honey bee, *B. terrestris* and *O. bicornis* have been established as test organism for field and semi-field tests. They are representative of social and solitary bee species, respectively. However, it might be necessary to consider additional test species in cases where effects on these two species cannot generally be extrapolated to all ecologically similar groups of bees (i.e. vulnerable groups). It is not possible to refer potential test species for other FVI taxa since there have been no published studies or test protocols.

Endpoints

As with sublethal laboratory effects, there are numerous ecological parameters that might be chosen as ecotoxicological endpoints. The above-mentioned studies investigated effects on e.g. immune response, foraging bouts and success, flower visitation, nest-building activity, colony growth, number of queens and workers, brood count and offspring sex ratio. As these effects may often be correlated it is important to choose a minimal number of ecologically meaningful endpoints which reflect changes that would lead to population declines (i.e. generational effects). Additionally, pollination should be assessed as an important ecosystem service of many FVIs. Furthermore, endpoints should at best also include measures of species interactions such as predation, parasitism, intra- and interspecific competition as well as indirect effects as discussed above. Generally, population preservation should be the main determinant for the choice of endpoints.

Test protocols

EFSA (2013) proposed test protocols to assess pesticide toxicity towards honey bees in field and semi-field studies. Furthermore, they suggested a semi-field and a combined field-to-laboratory approach for bumble bees and a field and semi-field approach for solitary bees. These test protocols could be adapted to incorporate generational effects (e.g. hatched offspring per *O. bicornis* female). The above-mentioned studies may also be used as templates (e.g. Peters et al. 2016, Rundlöf et al. 2015, Sterk et al. 2016). Test designs should be adequate to detect generational effects in the strict sense with high statistical power. Unfortunately, there are no study designs available at the moment to assess indirect, source-sink, biodiversity or species interaction effects.

Uncertainties

Despite their realistic setup field studies are often highly variable in their results and difficult to analyse using statistical tests. Because of cost and study plot restrictions (low replication), many field studies can only be analysed in a descriptive way. Semi-field studies have the advantage of still being performed in quite realistic settings but giving the option of more replicates and reduced variance. However, semi-field studies (especially tunnel studies) can hardly incorporate indirect, source-sink or species interaction effects due to reduced plot size and exclosure of the surrounding landscape and biota. It remains to be investigated if such ecological effects may increase pesticide effects on populations. Furthermore, there are only few FVI species that have been tested in field or semi-field studies. Moreover, the representativeness of these established test species for their own or even other FVI groups has not been sufficiently studied. Ecological trait differences (e.g. herbivore or florivore larval stages, soil or above-ground nester) may lead to substantial variation in population responses between FVI species. As there is not enough information available to address these issues at the moment, uncertainties should be reflected in the use of a conservative safety factor.

6.1.7 Conclusion

Research on toxic effects of pesticides has shown that they can impact wild bee species and cause lethal and sublethal effects, although the majority of recent studies was performed with neonicotinoids and honey bees. Laboratory investigations have shed some light on interspecific sensitivity differences and drivers of sensitivity. Furthermore, there is evidence of synergistic mixture toxicity and combined effects of pesticides and parasites. In field and semi-field studies such effects could also be shown. However, immune effects have only been investigated in honey bees, so far. The adverse influence of pesticides on reproduction, foraging and immune system was established in multiple studies at field-realistic doses. However, the wild bee study organism was usually the buff-tailed bumble bee *B. terrestris* and test substances usually neonicotinoid compounds. Information on toxic effects of other chemical classes on different wild bee species in realistic setups is still scarce. In that regard it is also imperative to evaluate the adequacy of test species as surrogates. There are test designs available to sufficiently investigate small-scale field effects on both species as proposed by EFSA (2013): *Bombus terrestris* is a generalist eusocial bee species which makes it a reasonable representative of the rather small and ecologically homogenous group of bumble bees (63 out of 1936 European species). *Osmia bicornis*, on the other hand, was not studied in such details in field and semi-field experiments. Therefore, there is insufficient information to assess field effects on this species and possibly on solitary bee species. Furthermore, *O. bicornis* cannot be representative for the large and ecologically heterogeneous group of solitary bees (1872 out of 1936 species). It is a polylectic ubiquitist cavity nester of medium size which is inherently different from all oligolectic, soil-nesting or smaller solitary bee species. Naturally, test organisms have to be easy to procure and to handle but to achieve biodiversity protection test species for ERA should also be chosen as representatives of ecological groups (please see 3.2 Categories derived from ecological traits).

6.2 Moths & butterflies

Caterpillars are the herbivore feeding stages of Lepidoptera (Scoble 1995) and, hence, they can cause damage also to crop plants. For this reason, insecticides against lepidopteran pests target predominantly caterpillars. Furthermore, studies on direct toxic effects (mortality) of insecticides on Lepidoptera have also focused on caterpillars (Kumar & Chapman 1984; Sinha et al. 1990; Davis et al. 1991a, b; Cilgi & Jepson 1995; Salvato 2001; Seljasen & Meadow 2006; de Jong et al. 2008; Dhingra et al. 2008; Hoang et al. 2011; Han et al. 2012; Hahn et al. 2015b; Pecenka & Lundgren 2015). However, in some studies, direct toxic effects of insecticides on adult Lepidoptera are also included (Salvato 2001; Hoang et al. 2011).

In addition to lethal effects, pesticides can also cause sublethal effects on Lepidoptera in different developmental stages. Sublethal reactions to insecticides include weight loss in caterpillars (Abro et al. 1993; Pecenka & Lundgren 2015), changes in caterpillar development and pupation times (Kumar & Chapman 1984; Pecenka & Lundgren 2015), changes in chemical communication and mating behavior of adult moths (Clark & Haynes 1992; Knight & Flexner 2007) and reduced reproduction of adult moths (Kumar & Chapman 1984; Abro et al. 1993; Knight & Flexner 2007; Han et al. 2012). Furthermore, unsuitable conditions during the caterpillar stage might also affect adult moths because there is a positive correlation between pupal weight and adult fecundity (e.g. Calvo & Molina 2005). Therefore, even if an insecticide does not have a lethal effect on a caterpillar, it might be detrimental to its further development and reproduction. Caterpillars have been observed to prefer untreated over insecticide-treated food (e.g. pyrethroids) (Kumar & Chapman 1984; Abro et al. 1993). Additionally, adult females can avoid oviposition on insecticide-treated surfaces (Kumar & Chapman 1984; Seljasen & Meadow 2006). In a semi-field study assessing the oviposition behaviour of *Hadena bicruris* moths, the females laid 40% fewer eggs on typical field margin non-target plants (*Silene latifolia*) sprayed with the insecticide lambda-cyhalothrin than on untreated control plants (Hahn et al. 2015b).

Next to insecticides, herbicides can also have negative effects on Lepidoptera. On the one hand, herbicides can reduce the occurrence of certain plant species (Schmitz et al. 2014a) which may also affect the availability of host plants for caterpillars and nectar plants for adult Lepidoptera. In addition, some herbicides have been observed to cause reduced flowering of plants (Schmitz et al. 2013, 2014b) which might further deplete nectar sources for adult Lepidoptera. On the other hand, herbicides can also influence host plant quality. Caterpillars of the cabbage moth *Mamestra brassicae* feeding on herbicide-treated *Ranunculus acris* plants showed reduced survival and delayed development (Hahn et al. 2014). These effects could be caused by changes in the nutritional value of the host plants (sulfonylurea herbicides inhibit the enzyme acetolactate synthase that is involved in the synthesis certain of amino acids (Drobny et al. 2012) or by herbicide effects on the photochemistry of plants through the production of secondary metabolites (Kjær et al. 2001).

Evidence of population declines caused by pesticides is scarce just as in bees: Forister et al. (2016) studied a possible link between neonicotinoid applications and California lowland butterfly populations. Using butterfly long-term monitoring data they could show that butterfly species richness is in decline since 1997 which is correlated with neonicotinoid usage in that area. Additionally, mostly smaller butterfly species were affected by the pesticides. Gilburn et al. (2015) investigated population indices of UK butterflies and found negative associations of 15 out of 17 studied species in areas with neonicotinoid usage.

Generally, toxic effects of agricultural pesticides have not been extensively studied in non-pest species. There is a need for more research focusing on common non-target species rather than pests and including moth species. Overall, sublethal effects have not been comprehensively assessed. There might be effects on larval development and metamorphosis as Hahn et al. (2014) showed first evidence for. Toxicity of pesticide mixtures or combinations with other stressors such as pathogens/parasites needs to be investigated. Furthermore, ecologically more relevant field and semi-field studies should be conducted to assess environmental effects on lepidopteran populations (Pisa et al. 2015; Wood & Goulson 2017). Suitable endpoints to start with are reproduction and foraging activity just as in bees. Furthermore, indirect effects on herbivore life stages should be evaluated which might also be of considerable importance to other FVI groups (e.g. beetles).

6.3 Tolerable/negligible effects

In FVI risk assessment the general protection goal should be the preservation of stable population levels as suggested by EFSA (2013, 2015). Several proxy parameters such reproductive success and foraging capacity can be used to assess population effects. However, it is difficult to define magnitudes of tolerable effects. EFSA (2013) used a honey bee colony population dynamics model devised by Khoury et al. (2011) to define a negligible population effect as a reduction in colony size by up to 7%. However, this model is not feasible to simulate population dynamics of bumble bees and solitary bees due to ecological differences. Bumble bees are more similar to honey bees than solitary bees because they also live in colonies but also have smaller colony size and quasi-solitary life stages when queens start a new colony in spring. Solitary bees do not have a social structure which means that their population dynamics and reaction to pesticide stress is completely different from honey bees. Using modelling approaches (on the landscape-scale at best) that consider ecological attributes, sensitive life stages and exposure of bumble bees, solitary bees and all other FVI groups (Table 31) it may be possible to derive magnitudes of tolerable population effects. In the meantime, a protective risk assessment can only be achieved by making conservative assumptions. Many FVIs are under threat, for example half of all German bee species are red-listed (Westrich et al. 2011). There is not enough data for all of Europe to make a qualified assessment but the situation may be similar. For most other FVI groups their threat status remains unclear. However, a recently published study showed a decline of more than 75% of flying insect biomass in 27 years in the agricultural landscape of Germany including mostly FVIs (Hallmann et al. 2017). A situation where the amount of flying insect biomass drastically declined or half of all species of a FVI group are endangered calls for strict regulation of all factors that have adverse effects on their populations. Therefore, in absence of quantitative reference data to assess what might be tolerable the only adequate precautionary action would be to allow for no population losses at all to protect ecosystem services and biodiversity.

6.4 Conclusion

Agricultural pesticide applications can result in lethal and sublethal effects on FVI species. This has been shown in laboratory studies mostly with honey bees but also other bee and some lepidopteran species. However, there is a need for further investigations of acute and chronic toxicity in wild bees, lepidopterans (especially moths) and species of other FVI groups with a focus on effects below lethal levels of pesticides. Effects of other pesticide classes than neonicotinoids, mixture toxicity and combined effects with other stressors such as parasites and pathogens should be investigated. Test species should be selected according to their ecological attributes as representatives of FVI (sub)groups. Furthermore, the impact of field-relevant pesticide doses should also be studied in realistic field and semi-field experiments using other FVI organisms than the honey bee or bumble bee species. This research program should incorporate toxic effects of pesticide product adjuvants, as well as the indirect effects of pesticides and their impact on ecosystem services of FVIs and their population responses. Moreover, field experiments should be designed to allow for at least some inference on source-sink dynamics. Otherwise, these effects should be simulated in landscape-scale models.

Open questions/research opportunities

- Other species than the honey bees (and bumble bees) should be tested, especially for sublethal and field effects.
- Herbicide effects on food plant quality as well as food plant presence and inflorescence need to be evaluated.
- Pesticide impact on FVI populations/biodiversity & ecosystem services needs further study.
- Ecologically adequate population models need to be developed to define tolerable effects on FVIs.
- Source-sink dynamics need to be incorporated in field study designs and landscape-scale model need to be developed.

Table 31: Overview of parameters that affect the exposure and effects of pesticides on life stages of bee and lepidopteran species.

FVI group	Life stage	Sensitivity (relative to other life stages)	Impact on population	Exposure route	Likelihood of exposure
Bees	Larvae	Unknown (possibly higher)	High in solitary and low in social species	Oral (might also be contact)	High (pollen, nectar, soil)
	Adults	Unknown (possibly lower)	High in solitary and low in social species (Exception: bumble bee and other primitively eusocial species' queens in quasi-solitary life stage)	Oral and contact	High (pollen, nectar, soil, leaf/stem material, extrafloral nectaries/honeydew, guttation, puddles, surface water)
Moths & butterflies	Larvae	Unknown (possibly higher)	Unknown	Oral and contact	High (stem/leaf material)
	Adults	Unknown (possibly lower)	Unknown	Oral and contact	High (pollen, nectar, soil, possibly extrafloral nectaries/honeydew, guttation, puddles, surface water)

7 Risk assessment

7.1 Recommendations for FVI risk assessment concept

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7.1.1 Protection goals

Apart from the ecological services "pollination", "education, inspiration and aesthetic value" and "food provision" defined as protection goals for NTAs by EFSA (see section 2.1), FVIs populations should be protected because of their inherent value. This goal is reflected at best by the protection goal "biodiversity/genetic resources" which is also outlined in the EFSA scientific opinion on risk assessment for non-target arthropods (EFSA 2015) (see also section 2.1). The EFSA further specified this protection goal according the guideline in the "Scientific Opinion on the development of specific protection goal options" for NTAs (EFSA 2015) (Table 32).

Table 32: Specific protection goal „biodiversity/genetic resources" as defined by EFSA (2015) for non-target arthropods according to the EFSA scientific opinion on the development of specific protection goal options.

Specific protection goal "biodiversity/genetic resources"	
In-field	
Ecological entity	(Meta)population
Attribute	Abundance
Magnitude/Temporal scale	Small effects (10-35%) on abundance and occupancy of NTA populations over months (max. 6 months) are tolerated. Note that the ecological entities are the populations and not functional groups.
Off-field	
Ecological entity	Population
Attribute	Abundance
Magnitude/Temporal scale	Negligible effects are tolerated, i.e. ► local scale: ≤ 10% or comparable non-detectable effects on NTA species abundance that are directly caused by exposure in the off-field habitat; ► landscape scale: negligible effects (≤ 10%) on NTA species abundance and spatial occupancy are tolerated. However, year-on-year decline in the abundance of species should not be observed.

As already outlined in section 6.3 there is currently no sufficient database for wild bees, solitary bees and other FVIs available to define magnitudes of tolerable effects. Modelling approaches might help to answer this question for single species, but require detailed information regarding ecology and sensitivity of the respective species (see also section 7.2). However, for many FVIs such detailed information is currently not available. Because of this data lack and the fact that many FVI species are currently already threatened, precautionary no long-term population losses of FVIs should additionally occur in off-field habitats in consequence of the application of pesticides (see section 6.3). This conclusion is also supported by EFSA (2015) who explicitly recommends that at landscape scale, a year-on-year decline in the species abundance should be excluded. However, whether the magnitude of tolerable effects recommended by EFSA (i.e. ≤ 10% effect on populations off-field, see also Table 32) would be sufficiently conservative to meet this level of protection for FVI populations with an appropriate margin of safety, cannot be answered based on the information currently available (see also section 6.3). With this respect further research is necessary. As long as no further quantitative reference data is available for FVIs the specific protection goals for FVI risk assessment in the pesticide authorisation process should be defined by regulators on the basis of the recommendations developed by EFSA for NTAs. However, this approach needs to be refined in the future based on experiences gained during the future authorisation process and/or as soon as adequate FVI specific data will be available.

In short: Protection goals for FVI risk assessment

- No long-term declines of FVI populations should occur in off-field habitats.
- However, there is currently no sufficient database for wild bees, solitary bees and other FVIs available to define magnitudes of tolerable effects which would be sufficiently conservative to meet this level of protection for FVI populations with an appropriate margin of safety.
- As long as no further quantitative reference data is available for FVIs:
- Specific protection goals for FVI risk assessment should be defined by regulators on the basis of available knowledge.
- A feasible approach might be an orientation on the recommendations developed by EFSA for NTAs (Table 32). However, this approach needs to be refined based on future experiences gained during the authorization process and/or as soon as adequate FVI specific data will be available.

7.1.2 Focal species and test species

EFSA noted 2015 in their scientific opinion addressing the state of the science on risk assessment of plant protection products for non-target arthropods that the implementation of the focal species approach for NTAs might be difficult, because of the great species diversity of arthropods and the highly diverse communities present throughout Europe. This might also be valid for the group of FVIs.

The EFSA recommended rather an approach based on ‘indicator groups’ that are representative for a set of several species with common trait combinations (EFSA 2015).

For **bees** a sufficient ecological database was available, and categories of bees with the same main ecological traits (mobility, lecty, sociality) were defined in section 3.2.1. Additionally, these categories were classified with respect to their vulnerability regarding the respective ecological traits (section 3.3.1). For each category a representative indicator species was proposed in section 0 resulting in a total of seven focal bee species. The focal species are representatives for the respective category.

The definition of focal species for **butterflies and moths** is quite more difficult due to the lack of data necessary to define groups based on ecological traits. Nevertheless, two butterfly species could be proposed as possible test species which were already used in Bt maize studies and for which some experience in testing is available. These species might be also suitable as focal species for a specific group (see also section 3.2.2.3). However, it has to be noted that these species do not represent all ecological groups of Lepidoptera and it is essential to gather more information with regard to butterflies and moth to support the selection of focal species. There are a few other butterfly species that might be chosen as test species. Two species that have been investigated in several studies are the large white *Pieris brassicae* and the small white *Pieris rapae* (Sinha et al. 1990; Davis et al. 1991; Cilgi & Jepson 1995; ICPPR 2017c). However, these species are most likely not suitable as focal species because of their traits assemblage leading to low population vulnerability.

The available data is not sufficient to propose focal species for moths. The sensitivity of the moth species *Helicoverpa zea* to pesticides has been investigated in laboratory tests and field efficacy studies. However - due to the traits and the possible resistance development - this species might neither be a suitable test species nor a suitable focal species for pesticide risk assessment.

Proposed FVI focal species / indicator species (bees) and test species (Lepidoptera)			
FVI group	**Focal species/test species**	**Ecological vulnerability**	**Remarks**
Bees	*Andrena viridescens*	High	Categories of bees with the same main ecological traits (mobility, lecty, sociality) were defined. For each category a representative focal bee species was proposed.
Bees	*Lasioglossum malachurum*	Medium	
Bees	*Hylaeus communis*	High	
Bees	*Nomada striata*	Medium	
Bees	*Andrena proxima*	High	
Bees	*Bombus terrestris*	Low	
Bees	*Osmia bicornis*	Low	
Lepidoptera	*Aglais urticae* or *Aglais io*	-	No sufficient data to define ecological sensitive groups. *Aglais urticae/Aglais io* were proposed as test species. They might also be suitable as focal species. More research necessary to support selection of focal/indicator species.

7.1.3 Relevant crops for risk assessment

In Germany the agricultural landscape is dominated by three main major crop types: arable crops, orchards and vineyards. The various crops within these groups vary in their attractiveness for FVIs (see section 3.1.2). For example, flowering crops (such as e.g. oilseed rape) are considered to be more attractive for FVI adults collecting nectar/pollen than wind pollinated crops (such as e.g. wheat). Crops identified by EFSA as attractive for adult bees are 3.1.2 (EFSA 2013a). However, as discussed in section 3.1.2 FVIs might also be present in non-flowering, less attractive crops, e.g. living there as larvae (Lepidoptera), or also FVI adults for nesting or sunning. Another reason for the presence of FVIs in non-flowering crops might be that FVIs are attracted by the crop via guttation. Based on available data it is currently not possible to rule out exposure to guttation droplets from certain crops (EFSA 2013). Therefore, exposure via guttation is assumed to be relevant for all crop types. Furthermore flower deposits might be relevant also in less attractive crops (e.g. wind-pollinated crops) in case that there are no other pollen/nectar resources in the surrounding. However, it might be assumed that in less attractive crops and crops harvested before flowering the probability of FVIs being exposed to pesticide residues might be lower in contrast to attractive crops.

Foraging on crop-associated wild plants, which might be present in the crop might lead to a relevant exposure. Particularly, when a high amount of crop-associated wild plants is available in a non-flowering crop, this originally as unattractive classified crop might become attractive for FVIs. This scenario is explicitly considered in the Draft Guidance on Risk Assessment to bees (EFSA 2013), where it is assumed that this exposure pathway might be relevant in case that the percentage of flowering weeds in the field exceeds 10%. However, it is currently not clear on which data the threshold value of 10% bases. Therefore, further research is necessary to determine the percentage of weeds in the field which lead to a relevant exposure of FVIs in an unattractive crop. As a pragmatic approach the threshold of 10% might be applied to FVIs as well, unless further data is available.

As a conservative approach it should be assumed that FVIs are potentially exposed towards pesticides on all fields and crop types, where pesticides are applied. Thus, an in-field pesticide risk assessment for pollinators should be obligatory for the authorisation of all pesticide uses. However, for the in-field risk assessment of uses in unattractive crops and crops harvested before flowering a refinement of exposure estimates in order to reflect the assumed reduced probability of exposure might be considered acceptable (Table 33).

Table 33: Crop types in FVI pesticide risk assessment.

Crop types	Unattractive crops	Attractive crops [1]	Harvested before flowering	Weeds in the field
Arable crops	x	x	x	x
Orchards	-	x	-	x
Vineyards	-	x	-	x
Risk assessment to be conducted	Yes [2]	Yes	Yes [2]	Yes

[1] Crops defined as attractive for bees by EFSA (2013).

[2] Refinement of exposure estimates in order to reflect the assumed reduced probability of exposure might be acceptable.

7.1.4 Exposure and risk assessment matrices

Relevant exposure scenarios for risk assessment of FVIs were discussed in section 5.3. For proposals to estimate these exposures please refer to section 5.4.

In the following, crop specific scenarios for pesticide risk assessment of FVIs are presented for the respective crops types (Table 33). The scenarios combine the developed exposure scenarios with potential effects on the respective life stage of the FVI groups, bees and Lepidoptera. The developed matrices shall provide guidance for the selection of relevant scenarios for the pesticide risk assessment for the intended specific pesticide use. In case that a scenario is defined as relevant (x), it has to be demonstrated that a pesticide use do not result in a non-acceptable risk for the respective species.

7.1.4.1 In-field, attractive crops

Table 34: Matrix for FVI risk assessment in-field of attractive crops (i.e. attractive, flowering arable crops, orchards or vineyards). x = relevant, scenario has to be addressed in risk assessment; - = not relevant, scenario has not to be addressed in risk assessment

Exposure scenario		Focal bee species		Focal lepidopteran species	
		Larvae	Adult	Larvae	Adult
Overspray / Dust dispersion		- / x[a]	x	x	x
Flower deposit (nectar/pollen)	Contact exposure	-	x	-	x
	Oral exposure	x	x	-	x
Stem/leaf deposit	Contact exposure	-	x	x	x
	Oral exposure	-	-	x	-
Soil deposit	Contact exposure [b]	x[c]	x	-	x
Systemic flower deposits (pollen/nectar)	Oral exposure	x	x	-	x
Systemic stem/leaf deposits	Oral exposure	-	-	x	-
Aqueous deposits	Guttation water	-	x	-	x[d]
	Surface water [e]	-	-	-	-
	Puddle	-	x	-	x

[a] Together with food collected by bee adults also contaminated dust might reach the brood cells and so the larvae might also be exposed. However, evidence for this exposure pathway in literature is currently lacking.

[b] Contact exposure via soil deposits is generally not well documented in literature. With this respect, further research is needed.

[c] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[d] Might potentially be relevant for Lepidoptera adults, however, there is currently no data available

[e] Exposure via surface water is not relevant for in-field assessment.

7.1.4.2 In-field, unattractive crops

Table 35: Matrix for FVI risk assessment in-field of unattractive crops (i.e. unattractive wind-pollinated arable crops). x = relevant, scenario has to be addressed in risk assessment; - = not relevant, scenario has not to be addressed in risk assessment

Exposure scenario		Focal bee species		Focal lepidopteran species	
		Larvae	Adult	Larvae	Adult
Overspray / Dust dispersion		- / x[a]	x	x	x
Flower deposit (nectar/pollen)	Contact exposure	-	x[b]	-	x[b]
	Oral exposure	x[b]	x[b]	-	x[b]
Stem/leaf deposit	Contact exposure	-	x	x	x
	Oral exposure	-	-	x	-
Soil deposit	Contact exposure [c]	x[d]	x	-	x
Systemic flower deposits (pollen/nectar)	Oral exposure	x[b]	x[b]	-	x[b]
Systemic stem/leaf deposits	Oral exposure	-	-	x	-
Aqueous deposits	Guttation water	-	x	-	x[e]
	Surface water [f]	-	-	-	-
	Puddle	-	x	-	x

[a] Together with food collected by bee adults also contaminated dust might reach the brood cells and so the larvae might also be exposed. However, evidence for this exposure pathway in literature is currently lacking.

[b] In general, flower deposits (nectar/pollen) might be relevant also in wind-pollinated crops in case that there are no other pollen/nectar resources in the surrounding.

[c] Contact exposure via soil deposits is generally not well documented in literature. With this respect, further research is needed.

[d] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[e] Might potentially be relevant for Lepidoptera adults, however, there is currently no data available.

[f] Exposure via surface water is not relevant for in-field assessment.

7.1.4.3 In-field, crops harvested before flowering

Table 36: Matrix for FVI risk assessment in-field of crops harvested before flowering. x = relevant, scenario has to be addressed in risk assessment; - = not relevant, scenario has not to be addressed in risk assessment

Exposure scenario		Focal bee species		Focal lepidopteran species	
		Larvae	Adult	Larvae	Adult
Overspray / Dust dispersion		- / x[a]	x	x	x
Flower deposit (nectar/pollen)	Contact exposure	-	-	-	-
	Oral exposure	-	-	-	-
Stem/leaf deposit	Contact exposure	-	x	x	x
	Oral exposure	-	-	x	-
Soil deposit	Contact exposure [b]	x[c]	x	-	x
Systemic flower deposits (pollen/nectar)	Oral exposure	-	-	-	-
Systemic stem/leaf deposits	Oral exposure	-	-	x	-
Aqueous deposits	Guttation water	-	x	-	x[d]
	Surface water [e]	-	-	-	-
	Puddle	-	x	-	x

[a] Together with food collected by bee adults also contaminated dust might reach the brood cells and so the larvae might also be exposed. However, evidence for this exposure pathway in literature is currently lacking.

[b] Contact exposure via soil deposits is generally not well documented in literature. With this respect, further research is needed.

[c] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[d] Might potentially be relevant for Lepidoptera adults, however, there is currently no data available

[e] Exposure via surface water is not relevant for in-field assessment.

7.1.4.4 In-field, weeds in the field

Table 37: Matrix for FVI risk assessment in-field of attractive crops (i.e. attractive, flowering arable crops, orchards or vineyards). x = relevant, scenario has to be addressed in risk assessment; - = not relevant, scenario has not to be addressed in risk assessment

Exposure scenario		Focal bee species		Focal lepidopteran species	
		Larvae	Adult	Larvae	Adult
Overspray / Dust dispersion		- / x[a]	x	x	x
Flower deposit (nectar/pollen)	Contact exposure	-	x	-	x
	Oral exposure	x	x	-	x
Stem/leaf deposit	Contact exposure	-	x	x	x
	Oral exposure	-	-	x	-
Soil deposit	Contact exposure [b]	x[c]	x	-	x
Systemic flower deposits (pollen/nectar)	Oral exposure	x	x	-	x
Systemic stem/leaf deposits	Oral exposure	-	-	x	-
Aqueous deposits	Guttation water	-	x	-	x[d]
	Surface water [e]	-	-	-	-
	Puddle	-	x	-	x

[a] Together with food collected by bee adults also contaminated dust might reach the brood cells and so the larvae might also be exposed. However, evidence for this exposure pathway in literature is currently lacking.

[b] Contact exposure via soil deposits is generally not well documented in literature. With this respect, further research is needed.

[c] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[d] Might potentially be relevant for Lepidoptera adults, however, there is currently no data available

[e] Exposure via surface water is not relevant for in-field assessment.

7.1.4.5 Off-field

Table 38: Matrix for FVI risk assessment off-field. x = relevant, scenario has to be addressed in risk assessment; - = not relevant, scenario has not to be addressed in risk assessment

Exposure scenario		Focal bee species Larvae	Focal bee species Adult	Focal lepidopteran species Larvae	Focal lepidopteran species Adult
Spray drift / Dust dispersion		- / x[a]	x	x	x
Flower deposit (nectar/pollen)	Contact exposure	-	x	-	x
	Oral exposure	x	x	-	x
Stem/leaf deposit	Contact exposure	-	x	x	x
	Oral exposure	-	-	x	-
Soil deposit	Contact exposure [b]	x[c]	x	-	x
Systemic flower deposits (pollen/nectar)	Oral exposure	x	x	-	x
Systemic stem/leaf deposits	Oral exposure	-	-	x	-
Aqueous deposits	Guttation water [d]	-	x	-	x[e]
	Surface water	-	x	-	x
	Puddles	-	x	-	x

[a] Together with food collected by bee adults also contaminated dust might reach the brood cells and so the larvae might also be exposed. However, evidence for this exposure pathway in literature is currently lacking.

[b] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[c] In worst-case larvae of soil nesting bees might be exposed. However, evidence in literature is currently lacking.

[d] Pesticide residues in guttation water off-field is currently not well documented in literature. Further research is necessary regarding this aspect.

[e] Might potentially be relevant for Lepidoptera adults, however, there is currently no data available.

7.1.5 Effect endpoints

To assess potential adverse effects of pesticide applications on FVIs, the measurement of effect endpoints are needed, which can be derived by toxicity tests. Standardised test protocols for FVIs are currently available for honey bees, bumble bees (*Bombus*) and *Osmia* (Table 39). Particularly for wild bees such as *Osmia* and bumble bees (*Bombus)* there was great effort in recent years to adapt honey bee tests on these species (e.g. (Hanewald et al. 2015)) and to develop standardised laboratory as well as field test methods (e.g. (Knaebe et al. 2017; Ruddle et al. 2017)). Test protocols are available for solitary bees and bumble bees (Table 39), but not all standardised test guidelines have been adopted on OECD level, yet. As outlined by EFSA (2015) standardised test protocols lack for Lepidoptera. However, proposals for test procedures can be found in open literature. For example, the proposed Lepidoptera focal species were tested already by Schuppener (2011) and used for Bt maize risk assessment (section 3.2.2.3).

Thus, in order to allow FVI risk assessment based on high quality and reproducible effect data the standardisation and/or adoption of test guidelines for the most relevant vulnerable FVI taxa should promoted with high priority. *Osmia bicornis* and *Bombus terrestris* (for both are test protocols available) were identified as focal bee species. However, it has to be noted that they are not representatives of the most vulnerable ecological bee categories (section 7.1.2, green box). Representatives of the most vulnerable bee groups are *Andrena viridescens, Hylaeus communis* and *Andrena proxima.* The development of tests for these most ecologically sensitive focal species is therefore recommended. However, until tests for these sensitive species are available, *Osmia bicornis* and *Bombus terrestris* in combination with an appropriate safety factor might be used in risk assessment.

Table 39: Overview of available test protocols of FVIs

Test protocol		Year	Endpoint
Honey Bees			
OECD Test No. 213 (OECD 1998b)	Honeybees, Acute Oral Toxicity Test	1998	LD_{50} oral
OECD Test No. 214 (OECD 1998a)	Honeybees, Acute Contact Toxicity Test	1998	LD_{50} contact
(Oomen et al. 1992)	Method for honeybee brood feeding tests with insect growth-regulating insecticides	1992	
ENV/JM/MONO(2007) 22 (OECD 2007)	No. 75, Guidance Document on Honey Bee (*Apis mellifera*) Brood Test under Semi-field Conditions, published in August 2007 in the series on Testing and Assessment	2007	mortality, flight activity, condition of the colonies, bee brood development
OECD Test No. 237 (OECD 2013)	Honey Bee (*Apis Mellifera*) Larval Toxicity Test, Single Exposure	2013	NOEC larvae
OECD draft (OECD 2014)	Draft Guidance Document on Honeybee *(Apis mellifera*) Larval Toxicity Test, Repeated Exposure	2014	NOEC larvae
(OEEP/EPPO 2010)	Side effects on honey bees. Section: Semi-field tests	2010	Mortality, behaviour/ foraging activity, colony condition (including brood status)
(OEEP/EPPO 2010)	Side effects on honey bees. Section: Field tests	2010	Mortality, behaviour/ foraging activity, colony condition (including brood status), additionally hive weight, pollen/nectar storage
OECD 245 (OECD 2017c)	Honey bee (*Apis mellifera* L.), chronic oral toxicity test (10-day feeding)	2017	LC_{50} (10 day), NOEC (mg/kg) for HPG
Draft EFSA Bee Guidance	Test to determine accumulative potential of the active substance	2013	
Bumble Bees			
Draft EFSA Bee guidance recommends to conduct OECD Test No. 213	Honeybees, Acute Oral Toxicity (Test should be adapted to Bumble bees)	1998	LD_{50} oral

Test protocol		Year	Endpoint
Draft EFSA Bee guidance recommends to conduct OECD Test No. 214	Honeybees, Acute Contact Toxicity Test (Test should be adapted to Bumble bees)	1998	LD_{50} contact
OECD 247 (OECD 2017b)	Bumble bee, Acute Oral Toxicity Test	2017	LD50 oral, NOED (if possible)
OECD 246 (OECD 2017a)	Bumble bee, Acute Contact Toxicity Test	2017	LD50 contact, NOED (if possible)
ICPPR draft (ICPPR 2017a)	Bumble bee Higher tier test	2017	brood development, colony weight, mortality, flight activity, reproduction
Solitary Bees			
Draft EFSA Bee guidance recommends to conduct OECD Test No. 213	Honeybees, Acute Oral Toxicity (Test should be adapted to solitary bees)	1998	LD_{50} oral
Draft EFSA Bee guidance recommends to conduct OECD Test No. 214	Honeybees, Acute Contact Toxicity Test (Test should be adapted to solitary bees)	1998	LD_{50} contact
ICPPR draft (ICPPR 2017b)	Mason bees (*Osmia* sp.), Acute Oral Toxicity Test	2017	LD_{50} oral, NOED (if possible)
ICPPR draft (ICPPR 2016)	Solitary bee, Acute Contact Toxicity Test	2016	LD_{50} contact
ICPPR draft (ICPPR 2017c)	Solitary bee, Higher Tier Test	2017	establishment (hatching rate, nest occupation) flight activity, reproduction, parasitation, immature mortality, foraging activity
Lepidoptera			
Schuppener (2011)	*Aglais io* or *Aglais urticae* Larval Toxicity Test (single or repeated oral exposure)	2011	LD_{50} oral, NOEC (feeding activity, biomass, development time (L3-L5), mortality)

An alternative approach to derive data for effect assessment even in case that no test method is available is the extrapolation based on already available endpoints. This is for example recommended by EFSA to derive toxicity values LC_{50} adult and NOEC larvae for solitary bees and bumble bees. However, potential differences in sensitivity across species are a source of uncertainty for a protective risk assessment. Thus, EFSA proposed to use the toxicity value available for honey bees applying a bridging factor of 10 (EFSA 2013a). A bridging factor of 10 as proposed by EFSA (2013) might be appropriate in many cases for the extrapolation of effect data from honey bees to solitary bees and bumble bees to cover uncertainties associated to potential differences in species sensitivity (Arena & Sgolastra 2014; Uhl et al. 2016; Heard et al. 2017). However, there are also indications that bridging factor of 10 might not be sufficiently protective for all pesticides. Uhl et al. (in prep.) tested the effect of 16 different insecticide formulations on *Osmia bicornis* and *Apis mellifera* (contact test). He found that in 13 out of 15 evaluable cases (87%) a bridging factor of 10 was protective also for *Osmia bicornis*. But in 2 of 15 cases (13%) *Osmia bicornis* was by a factor of 17 to 18 more sensitive than *Apis mellifera* (Table 30). Therefore, we recommend to conduct further research with respect to sensitivity differences of wild bee species (including especially species from the most vulnerable categories; please refer to section 7.1.2) and *Apis mellifera* towards wide spectrum of different pesticides (including various mode of actions etc.). Based on this data, a bridging factor might be derived which could be used for extrapolating from *Apis mellifera* to other species. In order to derive a scientifically sound bridging factor the data to be used should be high-quality and reliable data (i.e. following standardised test methods ideally conducted in compliance with GLP). However, as long as there is no such data available, the magnitude of the bridging factor should be defined by regulators on the basis of the available knowledge.

In case of Lepidoptera and moth it is not clear if extrapolating from honey bee toxicity data might be possible. Substance specific data to estimate the magnitude in sensitivity differences between honey bees and Lepidoptera species are currently not available. With this respect, further research (e.g. toxicity tests with Lepidoptera or literature research/data analysis) is recommended.

Bridging from honey bee endpoint to FVIs

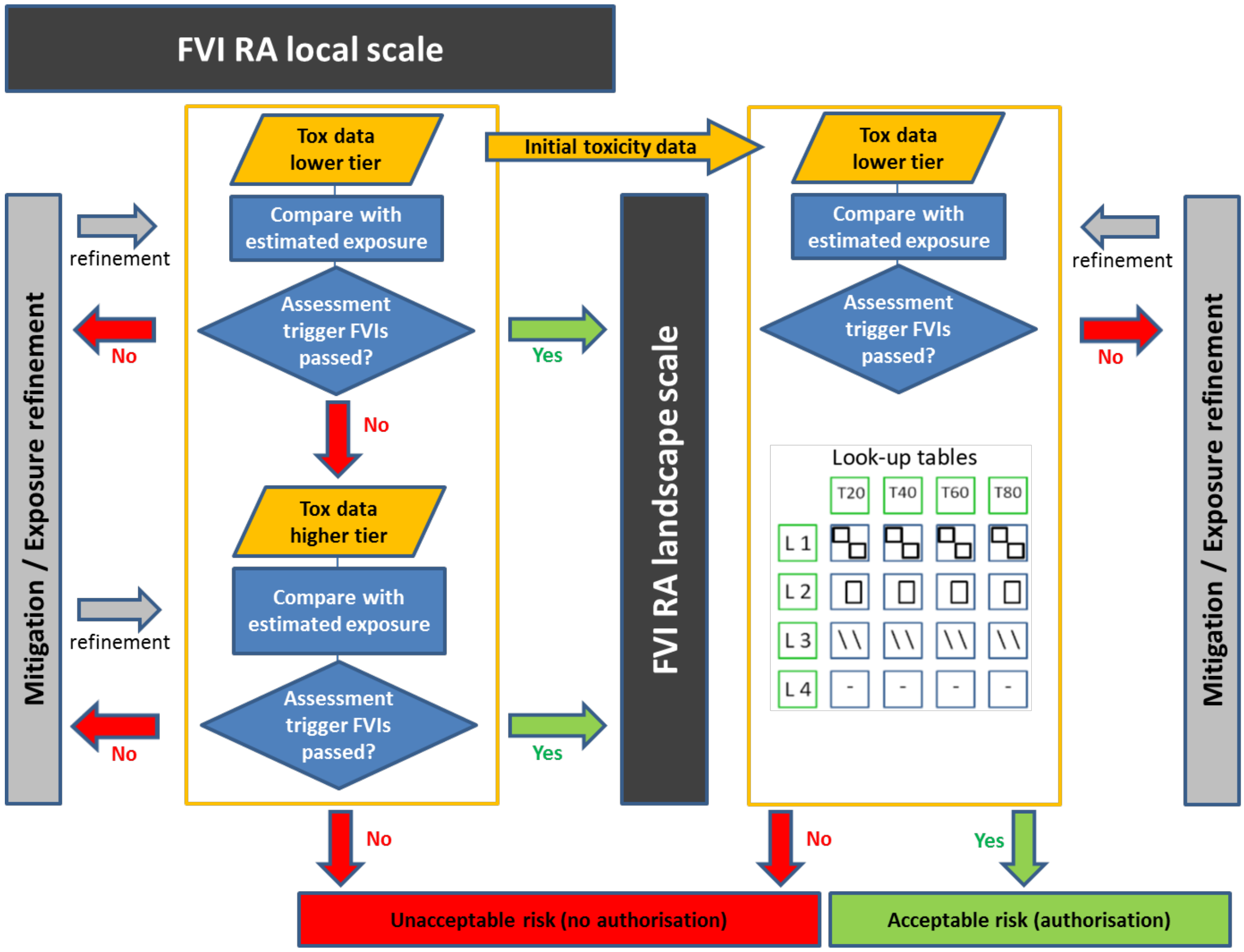
- Data endpoints for FVI effect assessment might be derived
  - by conducting toxicity tests or
  - by bridging from available endpoints to other endpoints.
- EFSA (2013) proposed a factor of 10 for bridging from honey bee endpoint to solitary and bumble bees. This might be appropriate in many cases; however, there are also indications that a factor of 10 might not be sufficiently protective for all pesticides.
- Therefore, we propose that the magnitude of the bridging factor should be defined by regulators on the basis of the available knowledge.

7.1.6 Risk assessment scheme

7.1.6.1 Overview

The EFSA (2015) proposed a new scheme for assessing the risk of NTAs towards pesticides. According to this proposal a local scale risk assessment as well as a landscape scale risk assessment should be conducted for species which show a high mobility. The risk assessment is only considered as safe, when both the local scale and the landscape scale risk assessment do not reveal unacceptable risks. Most FVIs are highly mobile and move between non target off-field areas and field areas. Thus, we propose to apply the concept of a risk assessment based on the local scale as well as on the landscape scale also for FVIs (Figure 21). For details of the different aspects of the proposed risk assessment for FVIs please refer to the sections below.

Figure 21: Risk assessment scheme proposed by EFSA (2015) for NTAs adapted to FVIs.



Source: own illustration, ecotox consult (adapted from EFSA 2015)

7.1.6.2 Local scale risk assessment Tier 1

Data requirements

As outlined in section 7.1.3 it should be assumed that FVIs are potentially exposed towards pesticides on all fields and crop types, where pesticides are applied. Thus, a risk assessment for pollinators should be obligatory for the authorisation of all pesticide uses. Based on the information on ecology and vulnerability currently available as well as the current available standardised test systems a basic toxicity data set should be required for lower tier risk assessment of a pesticide use:

- 2 bee species (larvae and adult), i.e. two of the proposed focal bee species.
- 1 Lepidoptera species (larvae and adult), i.e. *Aglais io* or *Aglais urticae.*

For <u>all crops</u> chronic effects should be assessed as part of the first tier in order to detect the occurrence of delayed effects of acute exposure. In accordance with the EFSA NTA opinion (EFSA 2015) we would recommend that effects on reproduction should be tested at tier 1. The tests should allow a robust prediction of adverse effects in the field.

However, at the moment standardised chronic lower-tier test protocols are neither available for FVI adults nor larvae (for an overview of available test protocols of FVIs please refer to section 7.1.5 Table 39). Thus, there is an urgent need to develop/standardise adequate test methods. As long as these test methods are lacking a pragmatic approach for tier 1 data requirements based on currently available test methods is proposed (Table 40).

Table 40: Pragmatic proposal for local scale tier 1 data requirements based on currently available test methods.

FVI Taxa	Exposure	No. of Species	Recommended Data Requirements	Pragmatic Proposal for Data Requirements Based on Currently Available Test Methods
Wild Bees (Adults)	Oral	2	Acute oral toxicity test	Acute oral toxicity test [f]
	Contact	2	Acute contact toxicity test	Acute contact toxicity test [f]
	Oral / Contact	2	Chronic toxicity test (reproduction) [a]	Surrogate: honeybee 10-d oral toxicity test adapted to wild bees [b, f]
Wild Bees (Larvae)	Oral	2	Chronic larval toxicity test (repeated oral exposure) [a]	Surrogate: honeybee larval toxicity test (repeated oral exposure) adapted to wild bees [b, f]
	Contact	2	Chronic larval toxicity test (contact exposure) [a]	- [c]
Lepidoptera (Adults)	Oral	1	Acute oral toxicity test [a]	- [d]
	Contact	1	Acute contact toxicity test [a]	- [d]
	Oral / Contact	1	Chronic toxicity test (reproduction) [a]	- [d]
Lepidoptera (Larvae)	Oral	1	Chronic larval toxicity test (repeated oral exposure) [e]	Larval toxicity test (repeated oral exposure)
	Contact	1	Chronic Larval toxicity test (chronic exposure) [a]	- [c]

[a] Standardised test protocols are not available yet.

[b] Due to the lack of standardised chronic test protocols, honeybee 10-day oral toxicity test and honeybee larval toxicity test (repeated oral exposure) should be used at this stage as surrogate for testing of adults and larvae, respectively. Tests should be preferably adapted to wild bees. If adaption to wild bees would not be feasible, bridging from endpoints derived from tests with honeybees might be acceptable.

[c] Due to the lack of standardised contact exposure test methods for larvae, it could be assessed case by case whether a bridging from endpoints from larval oral toxicity tests (repeated exposure) to the contact exposure test might be possible.

[d] Standardised test protocols are not available yet. In literature, some studies can be found in which direct toxic effects (contact exposure) on adult Lepidoptera were investigated (e.g. Salvato 2001; Hoang et al. 2011). Although the description of test methods lacks essential information, these publications might be used as a basis for the testing of adults as long as there are no standardised protocols available.

[e] Standardised test protocols are not available yet. However, proposals for test procedures can be found in literature (Schuppener 2011).

[f] Bridging from endpoints derived from tests with honeybees might be acceptable. To extrapolate from honeybees to wild bees a bridging factor of at least 10 should be applied to the honeybee endpoint. The magnitude of the bridging factor should be defined by regulators on the basis of the available knowledge.

Risk assessment

In tier 1 risk should be assessed for all contact and oral exposure pathways described in section 5.4 by relating expected exposure to toxicity.

Risk assessment - Contact exposure

Similar to the risk assessment of bees and NTAs, the calculation of a Hazard Quotient (HQ) for assessing the risk arising from contact exposure pathways is proposed. The risk is assessed by dividing the exposure estimate by the contact toxicity endpoint.

$$HQ = \frac{estimated\ contact\ exposure}{toxicity\ endpoint\ for\ contact\ exposure}$$

The higher the HQ, the higher is the expected risk. In the second step the HQ is then compared to a trigger value. For in-field risk assessment it is assumed that risk is acceptable when:

$$HQ_{in-field} = \frac{estimated\ contact\ exposure_{in-field}}{toxicity\ endpoint\ for\ contact\ exposure} \leq Trigger$$

Current NTA risk assessment uses a trigger value of 2. This value was derived according to a validation procedure comparing HQ values with (semi)field data for the species *A. rhopalosiphi* and *T. pyri* (SANCO 2002; EFSA 2015). For FVI risk assessment this trigger value has to be adapted. The trigger value has to assure that the protection goals are met.

For off-field risk assessment an additional assessment factor (AF) is included:

$$HQ_{off-field} = \frac{estimated\ contact\ exposure_{off-field}}{toxicity\ endpoint\ for\ contact\ exposure} * AF \leq Trigger$$

In the current risk assessment for NTAs the assessment factor is set to 10 (SANCO 2002). If the calculated $HQ_{off\text{-}field}$ is greater than 2, a higher tier risk assessment has to be performed. For FVI risk assessment trigger values and assessment factors should be adapted, in a way that the defined protection goals are met (please refer to section 7.1.6.3).

In a first step the intended application rate might be used for HQ calculation as a worst case assessment in tier 1. If trigger values are not passed a refinement of exposure estimates for each contact exposure scenario according to section 5.4 might be conducted.

Risk assessment - Oral exposure

For assessing the risk following oral exposure, an Exposure-Toxicity-Ratio (ETR) should be calculated, by dividing the exposure estimate (estimated for each oral exposure scenario, see section 5.4) by the toxicity endpoint for oral exposure:

$$ETR = \frac{estimated\ oral\ exposure}{toxicity\ endpoint\ for\ oral\ exposure}$$

Analogue to the contact risk assessment based on the HQ, the ETR is compared with a trigger value:

$$ETR_{in-field} = \frac{estimated\ oral\ exposure_{in-field}}{toxicity\ endpoint\ for\ oral\ exposure} \leq Trigger$$

For off-field assessment an additional safety factor should be included:

$$ETR_{off-field} = \frac{estimated\ oral\ exposure_{off-field}}{toxicity\ endpoint\ for\ oral\ exposure} * AF \leq Trigger$$

Trigger values and assessment factors have to be specifically adapted to FVI species (please refer to section 7.1.6.3).

In short: Tier 1 FVI risk assessment

Data requirements:

- toxicity data for 2 focal bee species (larvae and adult);
- toxicity data for 1 focal lepidopteran species (larvae and adult).

For all crops chronic effects should be assessed as part of the first tier in order to detect the occurrence of delayed effects of acute exposure. However, at the moment standardised chronic lower-tier test protocols are neither available for FVI adults nor larvae. As long as these test methods are lacking a pragmatic approach for tier 1 data requirements based on currently available test methods is proposed (please refer to Table 40).

Estimating the risk for FVIs from contact exposure pathways:

In-field acceptable risk if

$$HQ_{in-field} = \frac{estimated\ contact\ exposure_{in-field}}{toxicity\ endpoint\ for\ contact\ exposure} \leq Trigger$$

Off-field acceptable risk if

$$HQ_{off-field} = \frac{estimated\ contact\ exposure_{off-field}}{toxicity\ endpoint\ for\ contact\ exposure} * AF \leq Trigger$$

Estimating the risk for FVIs from oral exposure pathways:

In-field acceptable risk if

$$ETR_{in-field} = \frac{estimated\ oral\ exposure_{in-field}}{toxicity\ endpoint\ for\ oral\ exposure} \leq Trigger$$

Off-field acceptable risk if

$$ETR_{off-field} = \frac{estimated\ oral\ exposure_{off-field}}{toxicity\ endpoint\ for\ oral\ exposure} * AF \leq Trigger$$

Important to note: Trigger values and safety factors have to be specifically adapted to FVI species, so that defined protection goals are met.

7.1.6.3 Trigger values and assessment factors

To quantify the risk of the use of pesticides for FVIs, lower-tier toxicity data gained from standardised test protocols (e.g. acute oral and contact, chronic) need to be extrapolated to the ecosystem. Thus, trigger values and assessment factors are applied in risk assessment to lower-tier toxicity data. These trigger values and assessment factors should address all uncertainties associated with the extrapolation with an acceptable margin of safety. According to EFSA (2015) uncertainties comprises:

- inter- and intraspecies variation in sensitivity;
- potential difference between the application rate causing acute 50% mortality (LR_{50}) and the application rate causing other adverse consequences of concern (e.g. chronic, reproductive);
- potential difference between the application rate causing adverse effects in the laboratory and the application rate causing adverse effects in the field;
- effects under field conditions, e.g. effect from exposure of different life stages, indirect effects, effects on interactions, behavioral aspects, etc.

In order to meet the proposed specific protection goals for FVIs (please refer to section 7.1.1), the extrapolation (lower-tier to ecosystem) need to be calibrated by linking laboratory lower-tier toxicity data to effects in field studies for the same species. However, based on current available data a scientifically based calibration (including the determination of trigger values and assessment factors for FVIs) is not possible. With this respect further research is necessary.

7.1.6.4 Considering different ecological vulnerability categories of FVI-taxa

At least for bees, the available ecological database allowed to group bee taxa according to ecological species traits in categories and to classify these according to ecological vulnerability (please refer to section 7.1.2). For each category a representative focal bee species was defined. FVI risk assessment should assure the protection of all these groups of FVIs, also the most vulnerable ones.

As outlined in section 7.1.5 test protocols are only available for focal bee species *Bombus* and *Osmia*. However, these two species are not representatives of the most vulnerable ecological bee categories (please refer to section 7.1.2, green box). One approach to account for uncertainties associated with differences in the vulnerability between taxa could be the calibration of assessment factors and trigger values to the most ecological vulnerable taxa (please refer to the paragraph on trigger values and assessment factors above). Thus, risk assessment would be protective for the whole FVI community including vulnerable taxa.

7.1.6.5 Local scale risk assessment Tier 2

For all crops an intermediated refinement step at the local scale addressing uncertainties associated to potential differences in species sensitivity might be proposed. In analogy to the recommendations by the Aquatic Guidance Document (EFSA 2013b) a geomean approach and /or species sensitivity distribution (SSD) approach might be applied. However, these approaches would require test methods for a wide variety of FVI taxa. Thus, due to the current lack of standardised test protocols as described above neither the geomean nor the SSD approach are considered feasible for the time being. However, it should be decided by regulators on the basis of the best available knowledge whether such a refinement should be considered as acceptable and which level of protection should be applied.

7.1.6.6 Local scale risk assessment Higher Tier

In case an acceptable risk cannot be demonstrated in lower tiers the risk assessment might be refined in a higher Tier risk assessment at the local scale. By conducting adequate higher tier studies the applicant has to demonstrate that there is no unacceptable risk for vulnerable FVI species (outlined in section 3.3). Suitable approaches might be conducting semi-field/field effect studies to refine substance specific aspects or conducting exposure studies to refine pesticide residues in environmental matrices (e.g. nectar/pollen, plant, soil).

It is important to note that even if semi-field or field studies with FVI species are conducted to refine the risk estimation, a certain amount of uncertainty remains, which has to be addressed by an adequate safety factor. Possible uncertainties arise for example from:

- Interspecies variability: Particularly in case that the test was conducted with only one test species, there is high uncertainty arising from extrapolating from the test species on other species. Uncertainty might be reduced by field multi-species studies (Pettis et al. 2014).
- Intraspecies variability (Pettis et al. 2014).
- Temporal factors: Reaction of tests organisms might vary with seasons.
- Geographical/climatic factors: The results of a study conducted in a certain geographical region might vary in other regions in Europe (e.g. due to climatic conditions) (Pettis et al. 2014).
- Extrapolation from one crop to another (EFSA 2013a).
- Extrapolating from the (semi)-field study to real world, because of test conditions varying from conditions in real ecosystems.
- Year by year application might not be addressed by the higher tier study.
- Uncertainty within the study arising e.g. from precision of effects measurement, correctness of experimental conditions and parameters, occurrence of potential confounders, exposure assessment, statistical methods.

A guide how to analyse uncertainty within a study as a systematic tabular approach is described by EFSA (2013a). As recommended by EFSA (2013a) every refined risk assessment (i.e., higher tier risk assessment) should be accompanied at least by a qualitative evaluation of the uncertainties which might influence the risk (Table 41). Such an evaluation of uncertainties might help to determine a protective safety factor for higher tier risk assessment. The level of protection has then to be determined on the available knowledge by the competent regulatory authority.

Table 41: Tabular approach to assess uncertainties in higher tier studies as proposed by EFSA (2013).

Source of uncertainty	Potential to make true risk lower	Explanation	Potential to make true risk higher	Explanation
Concise description of first source of uncertainty	Degree of negative effect (e.g. ---)	Short narrative text explaining how this factor could make true risk lower		
Second source of uncertainty	n.r.	-	Degree of positive effect (e.g. +++)	Short narrative text explaining how this factor could make true risk lower
Add extra rows as required for additional sources of uncertainty	-	Note: many uncertainties may act in both positive and negative direction	+	
Overall assessment	Narrative text describing the assessor´s subjective evaluation of the overall degree of uncertainty affecting the assessment outcome, taking account of all the uncertainties identified above. The overall assessment should be a balanced judgement and not simply a summation of the plus and minus symbols.			

7.1.6.7 Landscape scale risk assessment

For populations of NTA species showing high mobility, the assessment of potential risks at the local scale will not protectively address adverse effects of pesticides applied in-field on the off-field population (EFSA 2015). Due to the high mobility of most vulnerable FVIs groups, we also propose to consider effects on FVIs at the landscape scale. At landscape scale modelling approaches (e.g. individual based models such as ALMaSS, Topping et al 2003) are needed to assess the risk with respect to spatio-temporal variation in pesticide dynamics and the interaction with spatial dynamics of mobile FVI. For more details on the proposed modelling approach please refer to section 7.2 (Landscape-scale population-level pollinator model feasibility).

As modelling is very complex and requires high effort, EFSA (2015) proposes a lower tier within landscape scale modelling by using so called look-up tables. Look-up tables shall provide the results of pre-run modelling scenarios, which can then be re-checked for the specific data of the applicant (e.g. environmental fate, GAP, intended use etc.). The look-up tables to be developed for NTAs as proposed by EFSA (2015) would need to be adapted with respect to the spatial dynamics of mobile FVI to meet FVI specific risk assessment requirements.

7.1.6.8 Mitigation of identified risks

In case that an acceptable risk cannot be demonstrated for a specific pesticide use by local and landscape scale risk assessment, the implementation of mitigation measures might be suitable to reduce risk for FVIs. Effective risk mitigation measures for protecting FVIs against effects of pesticides are evaluated in section 8.2 according to their efficiency (to reduce pesticide entries and to promote FVIs), feasibility and acceptability by farmers. Table 47 gives an indication which measures might be particularly appropriate to promote FVIs in the agricultural landscape. Especially mitigation measures aiming to affect FVI populations via change in landscape design might have great potential to reduce risk for FVIs over several years of pesticide applications. Such measures include the implementation of wildflower in-field buffers, conservation fallows or hedges. Such measures might be included in risk assessment as also proposed by EFSA (2015).

EFSA (2015) further outlines that there might also be synergistic effects with other regulatory frameworks such as the greening measures under the EU Common Agricultural Policy. As one main part the Greening includes the designation of ecological focus areas (EFAs) with the aim to maintain biodiversity and natural resources. The different types of EFAs were also evaluated with respect to their potential to promote FVIs (see section 8.4.2). Minimum requirements for the EFAs and additionally legally requirements were discussed in section 8.4.4 and 8.4.5.

The percentages of EFAs necessary to sufficiently compensate pesticide effects on FVI populations are difficult to determine based on the data currently available. Recent studies report that percentages of 3-7% (Cormont et al. 2016) of natural habitats or 7.5% (Holland et al. 2015) of uncropped land were appropriate for promoting FVIs (see section 8.4.4). Oppermann et al. (2016) showed that the implementation of 10% enhancement areas (such as flower strips, fallows) in a 50 ha study region promoted clearly wild bee abundance and diversity (Oppermann et al. 2016; Maus et al. 2017). Also within the project "Erweiterung des Risikomanagements durch Kompensationsmaßnahmen – Erfassung und Definition der Kompensationsleistungen geeigneter Managementmaßnahmen" it was concluded that a minimum proportion of 10% compensation measures of total arable land is needed to avoid the manifestation of indirect PPP effects at population level (Hötker et al. 2018; Swarowsky et al. 2018). Since currently only a few studies are available, further research is needed with regard to this topic. However, based on the few current available data a possible mitigation measure might be that "a certain pesticide use is allowed only in case that a certain amount of the agricultural land (e.g. 7%) in the surrounding account for EFAs". Thereby it is furthermore important that these EFAs are of high quality, i.e. those EFAs for which efficiency to promote FVIs was scientifically demonstrated

(Table 48 & Table 47). These were fallows, hedges, field margins, buffer strips or strips of eligible hectares along forest edges (at best seeded with wildflowers).

Possibilities to include effective mitigation measures in risk assessment

- Especially mitigation measures aiming to affect FVI populations via change in landscape design might have great potential to reduce risk for FVIs over several years of pesticide applications. Such effective measures include the implementation of wildflower in-field buffers, conservation fallows or hedges. These measures might be included in risk assessment.
- A possible point-based approach how such measures could be considered in risk assessment was developed within the project "Erweiterung des Risikomanagements durch Kompensationsmaßnahmen – Erfassung und Definition der Kompensationsleistungen geeigneter Managementmaßnahmen" (Hötker et al. 2018; Swarowsky et al. 2018). Measures were assigned with specific point scores representing their effectiveness. If 100 points per 100 ha were achieved, the application of a pesticide with a high risk for biodiversity might be still allowed.
- Ecological focus areas actually designed to maintain biodiversity and natural resources might have potential to reduce risk for FVIs from pesticide applications (e.g. wildflower in-field buffers or conservation fallows).
- A further possible approach to consider EFAs in the FVI risk assessment might be the introduction of the following risk mitigation measure in the pesticide authorization process: "A pesticide use is allowed only in case that a certain percentage of the agricultural land in the surrounding account for EFAs". The specific percentage of agricultural land in the surrounding has to be defined. Based on the available studies a value of 7% could be assumed. However, further research is necessary. Also the size of the area around the applied field which should be considered for identifying the required EFA quota has to be defined. Only EFAs should be considered where efficiency was scientifically demonstrated, e.g. fallows, hedges, field margins, buffer strips.

7.2 Landscape-scale population-level pollinator model feasibility

Christopher John Topping

7.2.1 Introduction to terrestrial landscape-scale population-level models for ERA

Landscape-scale population-level models (LSPLM) for ERA are not new. However, these are not common due to the difficulties of developing and2 maintaining the model systems needed to support these models. Smaller scale spatial models for population of animals have been developed, but here we differentiate these from dynamic landscape models that realistically simulate the pattern of land management and pesticide use on a spatial and temporal scale commensurate with animal behaviour leading to exposure and impacts at the individual level. In addition, they need to model populations large enough to be considered relevant for larger spatial areas in ERA.

EFSA reviewed one of the two current systems for modelling landscape-scale impacts of pesticides on pollinators ((PPR) 2015), the BEEHAVE model (Becher et al. 2014). They stated: "The modelling environment used by BEEHAVE (NetLogo) has an excellent user interface but provides limited opportunities for extending the model further development should use a standard, object-oriented language rather than NetLogo." The model was also criticised for a lack of a pesticide module and simplistic environment. In addition, the BEEHAVE model is specifically a honey bee single colony model and not suitable to model populations of individuals in a landscape. Therefore, BEEHAVE is not considered useful as a basis for constructing models of other pollinators.

Currently, the only models that satisfy all the population and landscape requirements are developed under ALMaSS (Topping et al. 2003), a C++ system of models designed to support simulation of terrestrial populations in managed environments. The current ALMaSS suite of models consists of:

- *Bembidion lampros* - a common carabid beetle, a generalist predator in arable fields (Bilde & Topping 2004);
- *Erigone atra* - a linyphiid spider, and a generalist predator in arable fields (Thorbek & Topping 2005);
- *Triturus cristatus* - Great crested newt, a widespread and threatened species in farmland (Topping et al in prep.)
- *Alauda arvensis* - European skylark, one of the most common open-field nesting birds, very often used as a key indicator of agricultural and pesticide impacts for farmland birds (Topping & Odderskaer 2004; Topping et al. 2013);
- *Perdix perdix* – Grey partridge, a common bird in European agricultural landscapes subject to heavy population declines (Topping et al. 2010a);
- *Microtus agrestis* - Field vole, one of the most common rodents in agricultural landscapes in northern-Europe, replaced by common vole further south (Topping et al. 2012);
- *Lepus europeaus* - European brown hare, a declining species common in agricultural landscapes (Topping et al. 2010b);
- *Oryctolagus cuniculus* - European rabbit, the most commonly used lagomorph in pesticide risk assessment, but not considered at risk of population decline) (Topping & Weyman, in prep).
- *Anser brachyrhynchus* – the pink footed goose, a population under trans-national management, of conservation interest and sometimes in conflict with farmers (Dalby et al, in prep);
- *Apis mellifera* – a honey bee model is under development under the auspices of EFSA's MUST-B working group.

Of these eight fully developed models, all except the partridge have been used to evaluate aspects relevant to pesticide risk assessment (e.g. Topping & Odderskaer 2004; Dalkvist et al. 2009; Dalkvist et al. 2013; Topping et al. 2014; Topping et al. 2015; Topping et al. 2016). These species are highly diverse but share some common characteristics that make them suitable for modelling at these scales and detail. These are that they are:

- well understood and researched, hence there is data to parameterise the models;
- their resources required to simulate their behaviour and ecology are predictable from mapping or other spatially explicit data;
- their dynamics are not closely linked to other species requiring a specific modelling of that species e.g. as would be the case if we attempted to model lynx and Canadian hare. Note that potentially this may be a problem if a pollinator species is closely linked in its population dynamics to a parasite (e.g. brood parasites and kleptoparasites in bees).

These attributes are therefore a prerequisite for any pollinator species considered for modelling at this scale and detail. Further specific requirements are specified under section 7.2.4 below.

These models are one part of the ALMaSS system, the other part that is required for the system to function is the simulation of the environment. Broadly, this serves the following functions:

- Represents a 2-dimensional map of the landscape (which can be up to 40 x 40 km in size), with a resolution of 1-m. This map effectively forms the environment in which the animals can exist and is comprised of a classification of all habitats relevant for the suite of animal species covered by ALMaSS.
- Within each habitat patch in the map environmental and biological characteristics are modelled and/or recorded as applicable. Environmental characteristics include soil type, area and size, pesticide concentrations, and external events (see below). Biological characteristics include

vegetation type, vegetation structure and growth, insect biomass, and spilled grain. Characteristics such as spilled grain are specific to particular animals, in this case geese. In fields weed and crop vegetation are simulated separately and in parallel.

- Pollen and nectar resources. Patch based methods for simulating the availability of pollen and nectar are under development for ALMaSS as part of a framework contract with EFSA for honey bee modelling. These will be available within the next two years.
- Provides daily or hourly weather data.
- Manages human activity, in particular farming (see Topping et al. 2016 for more details) but also management of e.g. roadside verges, and hunters and hunting. All farms are identified and the fields they manage allocated to the farm on the map. Farms are classified and crops allocated based on that classification. These crops are then rotated round the fields (if rotational). Each crop has a very detailed management scheme specifying all farming events (e.g. ploughing, sowing, harvest, pesticide applications), as a decision tree with conditions, probabilities and constraints for each decision. Once an event is triggered then the physical/environmental consequences of this event are modelled, e.g. if harvest, then the farmer will remove the vegetation from that field leaving stubble. These events are available to animals able to perceive them at that point in time and space. Management decision trees and events are detailed, e.g. in specifying the timing and amounts of any substance applied, as well as the vegetation or insect biomass impacts. This is particularly so for pesticides which are handled as a special case.
- Pesticides: Provides pesticide fate and environmental availability of pesticides. This is a large and complex part of the system, hence only an overview is presented here.
- For a pesticide that is sprayed and under consideration for ERA (typically only one pesticide is considered at a time for regulatory ERA): When a pesticide is sprayed by a farmer on a field the concentration of pesticide on plants and in soil is calculated per unit area based on the vegetation structure present at the time of spraying. This concentration will decrease with time following definable dissipation curves. When spraying drift is also taken into account also following user definable equations. Drift is affected by the assumed wind direction (daily variable). The areal units considered are also user definable down to a minimal resolution of 1m2. This means that after spraying events there is the potential to determine a pesticide concentration for any location in the landscape at any time.
- For non-ERA pesticides required to form a realistic background usage of pesticide and indirect effects. Following normal usage, the simulated pesticide is applied to a field (no drift) and any direct effects on animals or biological characteristics are calculated e.g. reduction in insect or weed vegetation biomass, or direct mortality to Bembidion if an insecticide is sprayed.
- Granular pesticide application and subsequent bioavailability is under development as is linking degradation rates in soil and vegetation to temperature and rainfall.
- Miscellaneous functions such as simulating traffic loads on roads, daylight hours and output facilities e.g. mapping crop usage.

To enable the landscape simulation part of ALMaSS certain data are required. These can be considered to be a prerequisite for any new area in which ALMaSS is expected to run. These data are specified under section 7.2.2.1 below.

7.2.2 General considerations on feasibility of landscape-scale population-level pollinator model development

The feasibility of developing a LSPLM for a pollinator will depend on the precise context defined for the model. There are three main aspects to consider:

- The geographical area modelled. Currently there are active models only for Denmark (total coverage), but landscapes for Norway, Poland and Portugal are under development. New landscape e.g. in Germany require the provision of data listed under section 7.2.2.1, as well as resources for developing a German-specific simulation (i.e. using the data needed).

- The detail of the modelling required. Although in the above it is assumed that the models represent daily life of the pollinators or other animals to a high degree of detail, this is not necessarily the case. For example if only the foraging behaviour is of interest, then models can be created where other functions (reproduction and mortality, and dispersal) are simplified or ignored. This will depend entirely on the goal specified for the model. In the rest of this document, we assume that the level of detail matches approximately the level used for the *Bembidion lampros* model (Topping et al. 2015). However, if less detail is required it will most likely increase feasibility.
- The regulatory scenarios needed. For example, it may be necessary to have multiple pesticides, multiple simultaneous exposure routes, specific crops and management regimes. The easiest most feasible will always be monoculture, single pesticide and single exposure route. Currently a single pesticide with others as background management (i.e. not directly part of the risk assessment) and realistic use and crop distributions is feasible and typical usage. However, depending on the exposure routes required, and the detail needed may require further development, and increase difficulty (see section 7.2.3).

One general consideration in planning any development of ALMaSS models is that once the landscape component has been developed for a region, then it is relatively little work to add existing or even new species. By far the greatest investment of resources is needed for the initial landscape set-up.

7.2.2.1 Information needed for development of landscape components

The methods for creating the landscape mapping and farming classification are presented in detail by Topping et al. (2016). The main data sources required for this for all realistic ALMaSS landscapes are as follows:

- Access to the EU subsidy support data submitted by farmers for EU subsidies under the CAP. This data shows both the ownership of fields each year and the types and areas of crops grown. The ideal format is a collated GIS layer identifying each farmer and his fields with an associated data file (e.g. Excel) showing each crop and the areas grown. This is typically the format that this data is reported to the EC. This data is collected by all EU countries where farmers claim CAP subsidies, however, access to the data varies with country. In Germany, access is controlled at the federal state level.
- Additional farm data on the number and type of livestock units present and whether the farm is organic or not. The availability of this data will depend on how animal numbers are registered in each country.
- Detailed information on management practices for each crop considered in the landscape.
- A source of typical pesticide use information per crop is required, either in the form of pesticide journals collected under the Sustainable Use Directive, or more likely via interviews with local farming advisors.
- GIS data layers covering non-field habitats in a much detail as is needed for the model use. In the case of pollinators, this needs to include woodland, gardens, roadside habitats and other semi-natural habitats (e.g. hedges, heathland). The aim here is that this data combined with a field map will provide an overall map with 100% coverage of the landscape at the detail required for the model purpose. Here again, local and national data sets determine the level and availability of detailed data available. At the European level there is coarse-grained data available such as CORINNE mapping, but typically higher quality data is available for local planning purposes including details of buildings, roads and non-agricultural habitats.
- Soil classification. Currently ALMaSS works with the following types: Lake, Sand dunes, Lavbund (no good English term, but an area in a valley bottom originally a wetland but now dry),

Marshland, Clay / Sand, Sand dunes, Moraine sand and gravel, Moraine clay and silt(clay soil), Sandy, Heavy Clay, Sand / gravel, Limestone, Chalk, Stoney.
- Weather data with daily precipitation, ground temperature, and wind run.
- Models for plant growth of primarily crops, but also generalised models for non-crop vegetation. There are models in ALMaSS that might be adapted but the extent to which this is possible will be determined by the difference in crop cultivars and climate compared to DK.

All of the above information is needed to make a realistic environment model. However, depending on the detail and ambition level desired the quality of some of this information can be relaxed. For example, farm classification may be of little importance as long as flowering crops can be reliably identified. The level of detail that can be left out is also determined by the target species. For example, honey bees have a wide range and can be modelled without fine grain details, whereas some bees e.g. *Osmia*, have a very restricted home range and therefore local scale details are more important.

7.2.2.2 General environmental information needed for development of pollinator models

For pollinators there is a particular need to model flowering resources in space and time (resource provisioning). The more realistically this is, the better the model. This is an extension currently being added to ALMaSS to enable the EFSA MUST-B honey bee model. The requirements in terms of data to enable this are as follows:

- Flowering phenology for crops and key non-crop resources for the area under consideration. This data may be found from literature sources and for most areas does not need to be collected in the field.
- A mapping of floral resources to the habitat types such that the flowering phenologies can be applied to patches. This is automatic for crops but requires either a fieldwork effort for a specific landscape or a statistical description of flowering resources per habitat type. In the latter case, each habitat patch must be given a flowering period and resource density, selected from a statistical distribution assumed for the landscape in question.
- For all species except honey bees there is need to be able to predict occurrence of suitable breeding locations. For some species, this may be done on the basis of vegetation, but others require soil type, slope and aspect to determine their nesting choices. These are not standard ALMaSS characteristics and may be difficult to map. This will affect individual model feasibility.

Assuming similar data is available from new (German) landscapes then there should be no technical issues with incorporating resource provisioning in new landscape developments.

7.2.3 Individual Exposure and Ecotoxicological Endpoints

The ecotoxicological endpoints that are available are very flexible in terms of what is possible in the model code. The limitations are typically toxicological knowledge leading to establishing effect and difficulties in realistically representing exposure.

7.2.3.1 Exposure

The basic ALMaSS outputs provide environmental pesticide concentrations per day in each 1-m of the landscape on vegetation. To express this as exposure to the individual requires the linking of movement behaviour and a method of expressing exposure. To do this movement must be modelled on a scale commensurate with the need for exposure modelling. For example, for animals that are present in the crop when it is sprayed a overspray calculation can be made to predict a dose based on surface area of the animal; their presence in the crop will be determined by the details and time frame of the movement modelled.

In the case of exposure by nectar or pollen, then the concentration of pesticide needs to be expressed in available pollen resources. Quantities of resources foraged and their subsequent consumption will thus determine the timing of exposure and the dose received.

Other sources of exposure such as contact or via drinking contaminated water, require specific sub-models and assumptions to be developed. They need to however, to work based on the pesticide concentrations in resources, on plants or in soil as the basic drivers. See section 5.4 for a detailed description of exposure routes and their quantification in a format suitable for modelling.

7.2.3.2 Toxic effects and endpoints possible

The most typical toxic effect possible is direct mortality, this is always possible assuming exposure can be calculated and mortality linked to a body-burden. However, depending on the detail in which the model is constructed other effects will be possible. In this case, the more detailed the model the greater the flexibility in representing endpoints. For example if reproductive details are represented explicitly in the model, then these mechanisms can be directly linked to pesticide effects e.g. in simulating growth rates of larvae, egg survival, egg fertility affected by the dose of pesticide received. Thus, toxicological effects can be expressed as interacting with any mechanism or entity that is explicitly modelled.

In all cases it is important to consider the problem of whether to use a stochastic death model or an individual threshold model for effects. Effects on the individual are based on the assumption that a given toxicological endpoint is measured over a test with a time component. For example we may have an LC50 measured over 7 days. The response to the pesticide is built into the model by assuming a threshold concentration above which there is a daily probability of mortality. This probability (p) is calculated from $(1\text{-m}) = (1\text{-p})^{d}$, where *m* is the proportion assumed to die (e.g. 0.5 for 50% mortality over the test period of 7 days) and *d* is the number of days over which the test was carried out. If an animal receives a dose above the trigger, then it is assumed to die with probability *p*. This approach is called the stochastic death model in GUTS TK/TD modelling (see Ashauer et al. 2015). This can be contrasted with the individual threshold approach, which sets an individual threshold above which death is certain. The implication of this choice is difficult to determine at the system level, but stochastic death has a larger probability of killing all exposed animals if multiple exposure occurs, whereas at low exposure levels the individual threshold approach leads to higher effects. It is also possible to include 'knock-on' effects for survivors, for example survivors may be more or less able to cope with subsequent exposure.

7.2.3.3 Endpoints

Here we define endpoints as an expression of the toxic effects at the population level. These can therefore be in terms of population numbers such as % mortality, change in population size, change in population distribution. Alternatively, these may be measures of processes individual characteristics such as per capita reproductive output, growth rates, lifespan. Since these are measured per modelled individual, any effect modelled at this level can be monitored and specific outputs designed. The endpoints possible are therefore a function of the choices made when implementing exposure and toxic effects, and any mechanism explicitly modelled in the species model can be made the subject of an endpoint if wished.

7.2.4 Feasibility of modelling key pollinator species

As a general rule, it is possible to model any species if we have some basic ideas of how they move, breed and die. However, for a model to be acceptable for risk assessment of pesticides there are some basic credibility criteria that need to be adhered to. One of these is that the data used to parameterise and design the model is scientifically supportable. Therefore, those species where there is little or no scientific literature support are not considered feasible for this approach.

For species modelling, individual-level behaviour in terms of dispersal, reproduction, mortality and individual/environmental interactions needs to be specified and parameterised. This should be describable at a temporal and spatial scale to be useful in differentiating impacts of pesticides and differing properties of pesticides. For example, annual modelling of population dynamics processes would not allow simulation of differing application schedules. Particular issues that need to be considered with pollinators over and above the standard modelling for any species are the interactions with the resource providing units (gain and removal of resources respectively), and therefore the foraging behaviour. Many invertebrates suffer from parasitic relationships, which if resulting in complex dynamics require modelling of both species. The details of biology and behaviour of the parasite may be obscure, and thus assumptions based on scarce data may be needed.

For bees, the list of species generated in section 0 has been used as the basis for feasibility. There are no obvious candidate species for modelling not present in this list.

7.2.4.1 *Andrena viridescens*

Research papers: A Web of Science search returned zero hits when *Andrena viridescens* was used as a topic keyword. When using the genus as a search term 290 references were returned of which 61 were classified as 'Ecology', for example a study on the population dynamics variability in a congeneric species (Franzen & Nilsson 2013). Some information is therefore available at a generic level but this is an otherwise poorly researched species. Specific requirements not known and details of reproduction, dispersal and foraging behaviour are likely to be difficult to find. Ecology and behaviour will therefore be very difficult to parameterise without conducting very detailed ecological studies.

Existing models: None known

Feasibility: Not feasible.

7.2.4.2 *Lasioglossum malachurum*

Research papers: A Web of Science search returned 42 hits when *Lasioglossum malachurum* was used as a topic keyword, and 325 when only *Lasioglossum* was used. Nine papers are classified as 'Ecology' of which one was a molecular study, and the eight ecology papers deal with eusociality, primarily from the queen's perspective. Some information is therefore available for this species, but very little on its ecology and behaviour excluding its unusual social structure. Ecology and behaviour will therefore be very difficult to parameterise without conducting very detailed ecological studies.

Existing models: None known

Feasibility: Not feasible.

7.2.4.3 *Hylaeus communis*

Research papers: A Web of Science search returned three hits when *Hylaeus communis* was used as a topic keyword none of which useful for model parameterisation, and 92 when only *Hylaeus* was used. 18 papers are classified as 'Ecology' at the genus level, of which most relate to Hawaiian species. The European species therefore seem to have been largely ignored in research literature. Ecology and behaviour will therefore be very difficult to parameterise without conducting very detailed ecological studies.

Existing models: None Known

Feasibility: Not feasible.

7.2.4.4 *Nomada striata*

Research papers: A Web of Science search returned zero hits when *Nomada striata* was used as a topic keyword, and only 41 hits when the genus was used as the search term. All papers returned under the genus search were taxonomic or survey-based. As such there is little or no data available in the reviewed literature on which to parameterise a model or describe the species behaviour.

Existing models: None known.

Feasibility: Not feasible.

7.2.4.5 *Andrena proxima*

Research papers: A Web of Science search returned three hits when *Andrena proxima* was used as a topic keyword. When using the genus as a search term 290 references were returned of which 61 were classified as ‘Ecology’, for example a study on the population dynamics variability in a congeneric species (Franzen & Nilsson 2013). Some information is therefore available at a generic level but this is an otherwise poorly researched species. Specific requirements not known and details of reproduction, dispersal and foraging behaviour are likely to be difficult to find. Ecology and behaviour will therefore be very difficult to parameterise without conducting very detailed ecological studies.

Existing models: None known

Feasibility: Not feasible

7.2.4.6 *Bombus terrestris*

Research papers: A Web of Science search returned 1676 hits when *Bombus terrestris* was used as a topic keyword, and 3204 when using *Bombus*. All aspects of ecology and behaviour are represented, including toxicology of some pesticides. The main challenge with this species is the citing of nest locations. Nests are located in existing soil cavities, e.g. a dry old mouse hole, and thus the occurrence of these in the landscape needs to be simulated. There is no obvious way this can be done except as a statistical process related to soil type and likely humidity. One other complication with this species is the impact of *Bombus bohemicus* and *B. vestalis*, a generalist and specialist brood parasite respectively. This relationship is quite well researched, however, details of the population dynamics resulting from these interactions may require further data.

Existing models: There are a number of existing models dealing with different aspects of bumble bee ecology (Cresswell et al. 1995; Raine et al. 2006; Stokes et al. 2006; Forrest & Thomson 2009; Rands & Whitney 2010; Dyer et al. 2014; Cresswell 2017). A combination of information and approaches taken in these models forms a good basis for the development of the detailed landscape-scale population-level models considered here.

Feasibility: Feasible.

7.2.4.7 *Osmia bicornis*

Research papers: A Web of Science search returned 113 hits when *Osmia bicornis/rufa* was used as a topic keyword, of which 18 were classified as ‘Ecology’. *Osmia* alone returned 505 results of which 87 were classified as ‘Ecology’. Foraging behaviour, physiology and reproduction are covered in the literature. There is also information on pesticide impacts. *Osmia bicornis* has been in focus as a reared pollinator for commercial use and therefore there has been an increase in research on this species. This species nests in hollow stems and as such simulation of its nesting site can be achieved from the current simulation of vegetation type and growth within ALMaSS.

Existing models: One known specific model (Ulbrich & Seidelmann 2001), an individual-based population model suitable as a basis for building a more comprehensive model. Development of a model in ALMaSS for this species has been initiated as a PhD study in Poland, Jagiellonian University of Krakow.

Feasibility: Feasible.

7.2.4.8 Butterflies

There are a number of butterfly species potentially suitable as examples of day-flying Lepidoptera for risk assessment of pesticides. Two species were suggested as potential focal species in this report: Small tortoiseshell (*Aglais urticae*) and European peacock (*Aglais io*). However, these species were selected for reasons of potential data availability and not because they are particularly representative of lepidopteran species in the agricultural environment.

From the point of view of landscape-scale ERA, the key selection factor for a model is also the availability of data. Here we can identify a few species where data is plentiful (*Pieris* spp., *Vanessa* sp. and *Aglais urticae, Polyommatus icarus,* and *Maniola jurtina*); in addition to data, we need to be able to model the availability of the larval food plant from mapping information. This removes many species from consideration (including *Pieris* spp. and *Helicoverpa*) leaving those that feed on grasses and common species such as stinging nettle (*Urtica dioica*). Of these, the Nymphalidae species are highly mobile and therefore not ideal for risk assessment purposes where spatial location is a factor. Thus, the only species that appears to fulfil all the criteria to be useful from a modelling perspective is *Maniola jurtina* (the meadow brown). This is a widespread species in Europe, dispersal ability is average with indications that genetic exchange between islands in close proximity is limited (Dowdeswell 1962; Ford 1964). It is a very common butterfly in farmland and has been extensively researched since first being the subject of genetics studies in the 1960 (Ford 1964). Dispersal and reproduction are well studied and the species is suitable for modelling on that basis. There is however, a potential for some difficulties in that as with almost all common butterfly species, the Meadow Brown is heavily affected by parasitoids, in this case a parasitic wasp *Apanteles tetricus*. Approximately 30-40% of larvae succumb to this parasitoid and the timing of peak impact is variable (Dowdeswell 1962). It is therefore necessary to model this parasitoid, and that is less feasible than modelling the butterfly alone.

7.2.5 Conclusions

The implementation of pollinator models in general will be eased with the developments currently being undertaken by EFSA for honey bees. This will provide models for resource production and pesticide exposure in nectar and pollen that could be used for future population models.

Development of models for the following species are considered feasible:

Bombus terrestris – the majority of data is easily accessible and with the exception of detailed pesticide data. Foraging, reproduction, breeding site choice are well described. Probably the main difficulty in modelling is the simulating nesting-site selection behaviour. These bees typically nest in existing cavities in the soil (e.g. old mouse holes), hence some proxy for availability of such structures will need to be found if the spatial distribution of nests is to be considered. Simulation of individual colonies, foraging and risk to pesticides would appear to be very feasible. Density-dependence may be a function of both suitable nest locations and the interactions with brood parasites such as *Bombus bohemicus* (cuckoo bee), and in both cases will require some assumptions to be made. However, there is basic data on parasite incidence that can be used as a basis for initial modelling.

***Osmia bicornis* –** somewhat less is known about this species compared to *Bombus terrestris*, but most information is available to some extent. Like *Bombus*, geo-locating reproduction sites may be somewhat problematical, but since this species uses hollow stems to nest the probability of finding stems may be linked to the predicted vegetation structure of the habitat patches. In this case determining density dependence may be difficult. Cleptoparasites are potentially and important driver for *Osmia,* and although there is some research available, the dynamics of these interactions is not well described.

Maniola jurtina – The meadow brown butterfly is feasible to model, but assumptions about the interaction with the parasitoid population of *Apanteles tetricus* will be needed. All other parameters and processes needed have been researched to a greater or lesser extent and exist in scientific publications.

This selection of species together with the honey bee would represent some key pollinator groups and ecologies. In particular, colonial to solitary bees are represented, as well as very local to migratory dispersal, and differing ranges of daily movement, from local to long range (up to 20 km for honey bees). Naturally, it is not possible to know everything about these species at this stage, but there is enough information on these species to keep unsupported assumptions in the model to an acceptable level. In all three cases the largest unknown will be the potential for parasite-driven population dynamics.

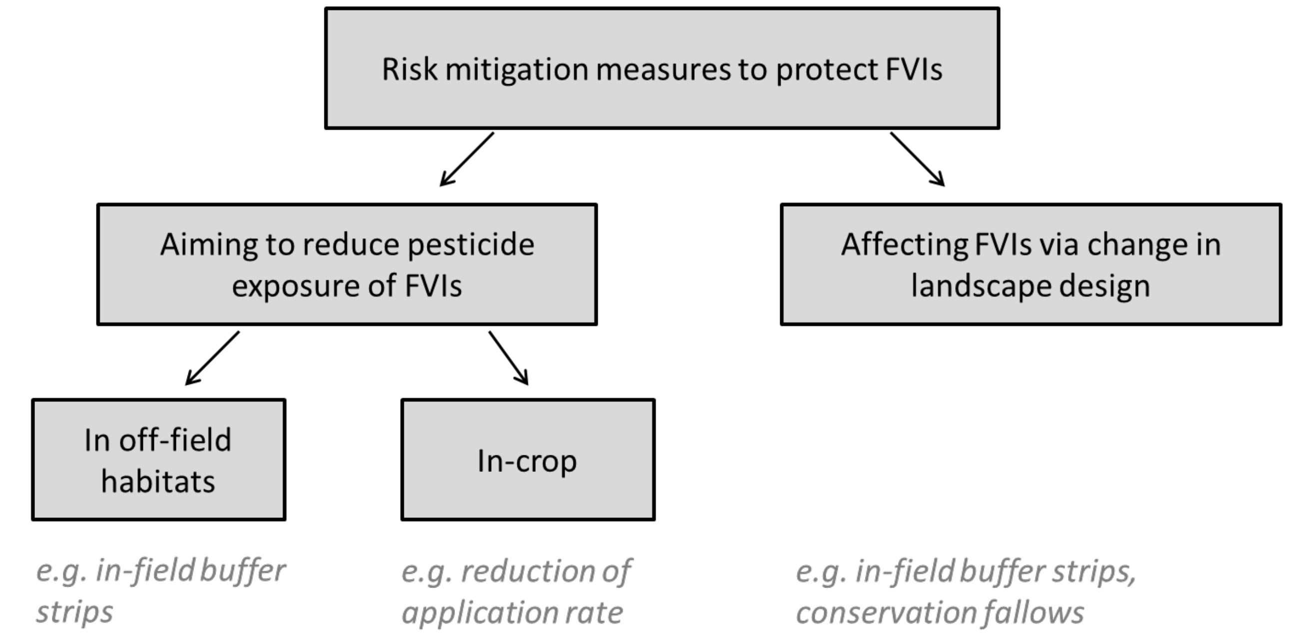
8 Risk management

Renja Bereswill, Kevin Krichbaum, Kristina Schmidt, Michael Meller

8.1 Risk mitigation measures for the protection of flower-visiting insects

For the protection of FVI against the effects of pesticides, risk mitigation measures can be implemented. In general, application-related measures which are related to the pesticide application process and landscape-related measures which require a change in the landscape structure can be differentiated (Bereswill et al. 2014). Measures can have the aim to reduce pesticide entries in off-field non target areas (e.g. field margins, hedges) and consequently to reduce the pesticide exposure of FVI in these areas (Figure 22, Table 42). Moreover, there are also measures which (additionally) aim to reduce the exposure of FVI in-crop[1] (e.g. reduction of pesticide application rate, pesticide application in the evening when diurnal FVIs are not active). Landscape-related measures have (additionally) the potential to promote populations of FVI in the agricultural landscape, because the change in landscape design leads to an enhancement in habitat availability and/or quality for FVI. Thus, pesticide effects on FVI might be compensated/reduced by these landscape-related measures.

Figure 22: Classification of risk mitigation measures for the protection of flower-visiting insects (FVI).



Source: own illustration, ecotox consult

The risk mitigation measures which are currently in force in Germany for the protection of terrestrial off-field habitats and their communities from the effects of pesticides are related to the pesticide application process (i.e., use restrictions). These measures include the use of drift reducing techniques and no-spray zones, which are (if necessary) applied in the pesticide authorization process for a specific pesticide use (Hahn et al. 2015). However, in Germany use restrictions are only mandatory when applying a pesticide neighbouring a terrestrial off–field habitat with a width of 3 m or more. A terrestrial off-field area with a width smaller than 3 m is currently generally not protected by any use

[1] For further information with regard to the differentiation between off-field habitat and in-crop area please refer to Figure 4.

restriction. Moreover, in regions with a sufficient percentage of small structures (e.g. hedges, small woodlands, riparian buffer strips) within the agricultural landscape (the specific regions/communes are listed in the German “Verzeichnis regionalisierter Kleinstrukturen”) certain use restrictions have not to be followed or other moderated use restrictions are valid (Hahn et al. 2015).

As a risk mitigation measure for the protection of bees, use restrictions are applied in the German pesticide authorization process to pesticides which are classified as hazardous for bees. Pesticides classified as hazardous for bees with the label B1 are not allowed to be used on plants which are in flowers or which are visited by bees (BVL 2016). Pesticides classified as hazardous to bees with the label B2 are also not allowed to be used on plants which are in flowers or which are visited by bees except after 11 p.m. when daily bee flight is finished (BVL 2016).

In addition to the risk mitigation measures currently in force in Germany, there is a wide spectrum of measures which might be suitable for the protection of FVI from the effects of pesticides. An overview of proposed mitigation measures is presented in Table 42.

Table 42: Overview of risk mitigation measures proposed for the protection of flower-visiting insects (FVI) from the effects of pesticides.

Risk management measure	Reduction of pesticide entries in off-field habitats	Reduction of pesticide exposure of FVI in-crop	Landscape design[1]	References[2]
In-field buffer strips	x		x	(Candolfi et al. 2000; EFSA 2015; Hahn et al. 2015)
Extension of small field margins	x		x	(EFSA 2015; Hahn et al. 2015)
Creation of conservation fallows	x		x	(EFSA 2015; Hahn et al. 2015)
High vegetation (hedges; tree rows)	x		x	(Candolfi et al. 2000; SANCO 2002)
Reduction of application rate	x	x		(SANCO 2002)
Application frequency and intervals	x	x		(SANCO 2002)
No-spray zones[3]	x	(x)[4]		(SANCO 2002)
Spray drift reducing techniques[3]	x			(Candolfi et al. 2000)
No overspraying of off field habitats	x			(Hahn et al. 2015)
Timing of application:				(SANCO 2002)
► application in the evening after honeybee flight[3]		x		
► no application when crops flowers or flowering vegetation between fruit		x		

Risk management measure	Reduction of pesticide entries in off-field habitats	Reduction of pesticide exposure of FVI in-crop	Landscape design[1]	References[2]
tree/vineyard rows are present[3]				
► application at low wind speeds	x			
Preservation and management of off-field habitats:				(Hahn et al. 2015)
► Sowing of seed mixes			x	
► Mowing rhythm			x	
► Maintenance of hedges			(x)[5]	
► Creation of nesting possibilities for bees e.g. bee			(x)[5]	(Plattform Bienenzukunft 2016; LTZ 2017)
Leaving deadwood in fruit orchards			(x)[5]	(Plattform Bienenzukunft 2016)
Sowing of seed mixes between vine and fruit tree rows			x	(Plattform Bienenzukunft 2016; LTZ 2017)

[1] A change in landscape design leads to an enhancement of habitat availability and/or quality for FVI. Thus there is potential to reduce pesticide effects on FVIs.

[2] Example of references which propose the mitigation measure as possible for the protection of FVI and arthropods, respectively.

[3] Currently in force in Germany.

[4] Reduction of pesticide exposure of FVIs in crop at least within the region of the no-spray zone.

[5] Measure might positively affect FVI populations in agricultural landscapes; however, an evaluation of this effect cannot be performed based on the current available scientific database.

8.2 Evaluation of proposed risk mitigation measures

8.2.1 In-field buffer strips

8.2.1.1 Reduction of pesticide entries

In-field buffer strips are defined as cropped (e.g. conservation headland) or uncropped strips (e.g. flowering strips) of a certain width at the edges of fields where pesticides are not applied. The creation of in-field buffer strips leads to an increase of the distance between the sprayed area of the field and the adjacent off-field non-target habitat. Consequently, also pesticide spray drift entries in the off-field area are reduced. The amount of pesticide drift in non-target habitats decreases exponentially with increasing distance (Ganzelmeier et al. 1995; Rautmann et al. 2001; Van de Zande et al. 2015). Thus, pesticide reduction efficacy of in-field buffer strips is dependent on their width. Pesticide reduction efficacy can exemplarily be calculated based on the German basic drift values (Rautmann et al. 2001) for different buffer width. According to such a calculation 5 m wide buffers adjacent to field crops might provide a 79% pesticide spray drift reduction in comparison to the pesticide deposition which is expected at a distance of 1 m. In case of adjacent fruit crops and vineyards, respectively, 5 m wide buffers might provide 32-47% and 55% pesticide spray drift reduction in comparison to the pesticide deposition expected at a distance of 3 m; higher efficacies of 60-77% and 85% can be reached by 10 m wide buffers adjacent to fruit crops and vineyards, respectively.

Besides their width, pesticide reduction efficacy of buffer strips is influenced by the vegetation type (present on the buffer). On the one hand, the presence of hedges or trees improves the potential of the buffer to efficiently reduce spray drift (see also section 8.2.4). On the other hand also low vegetation such as grasses or flowers can potentially reduce pesticide spray drift entries in comparison to bare ground (Bereswill et al. 2014). Canopy roughness and occurring filtering effects influence the drift particles retention process (Koch et al. 2003). Wolf et al. (2004) investigated a 15m-wide grassed riparian buffer strip (height of grass vegetation: 0.75 m): in comparison to a 15m-wide buffer strip consisting of bare soil spray drift was reduced by 50-60%. Van de Zande et al. (2000) noticed for a 1.25 wide buffer strip with a *Miscanthus* sp. vegetation of height of 0.5 m and 1 m reduction efficacies of 50% and 80%, respectively.

8.2.1.2 Effects on FVI

Via the reduction of pesticide entries in off-field habitats also the pesticide exposure of FVI present in these habitats is reduced. Moreover, in-field buffer strips have the potential to positively affect FVI populations in the agricultural landscape via an enhancement of habitat availability and/or quality (Table 42).

Several field studies can be found in literature investigating the effects of different in-field buffer types on FVI abundance and species richness (Table 43). The research is focused on the effect of in-field buffers on bees (8 studies) and butterflies (5 studies). Only three studies investigated potential effects on hoverflies. The main results of these research studies are presented in Table 43.

In general, there are four different types of in-field buffer strips investigated: conservation headlands (unsprayed field edges), strips with natural regenerated vegetation, grassed buffer strips and strips seed with wildflower or nectar and pollen mixture. A total of 25 in-field buffer strip treatments[2, 3] were investigated in all studies presented in Table 43 (Figure 23). Nineteen studies showed positive effects on FVI abundance and/or diversity (11 x wildflower strips, 4 x grassy strips, 1 x natural regenerated strip, 3 x conservation headland). In six studies no effect (2 x grassy strips, 2 x natural regenerated strips, 2 x conservation headland) could be found on FVI abundance or species richness. Not a single study demonstrated negative effects of in-field buffer strips on FVIs (Figure 23).

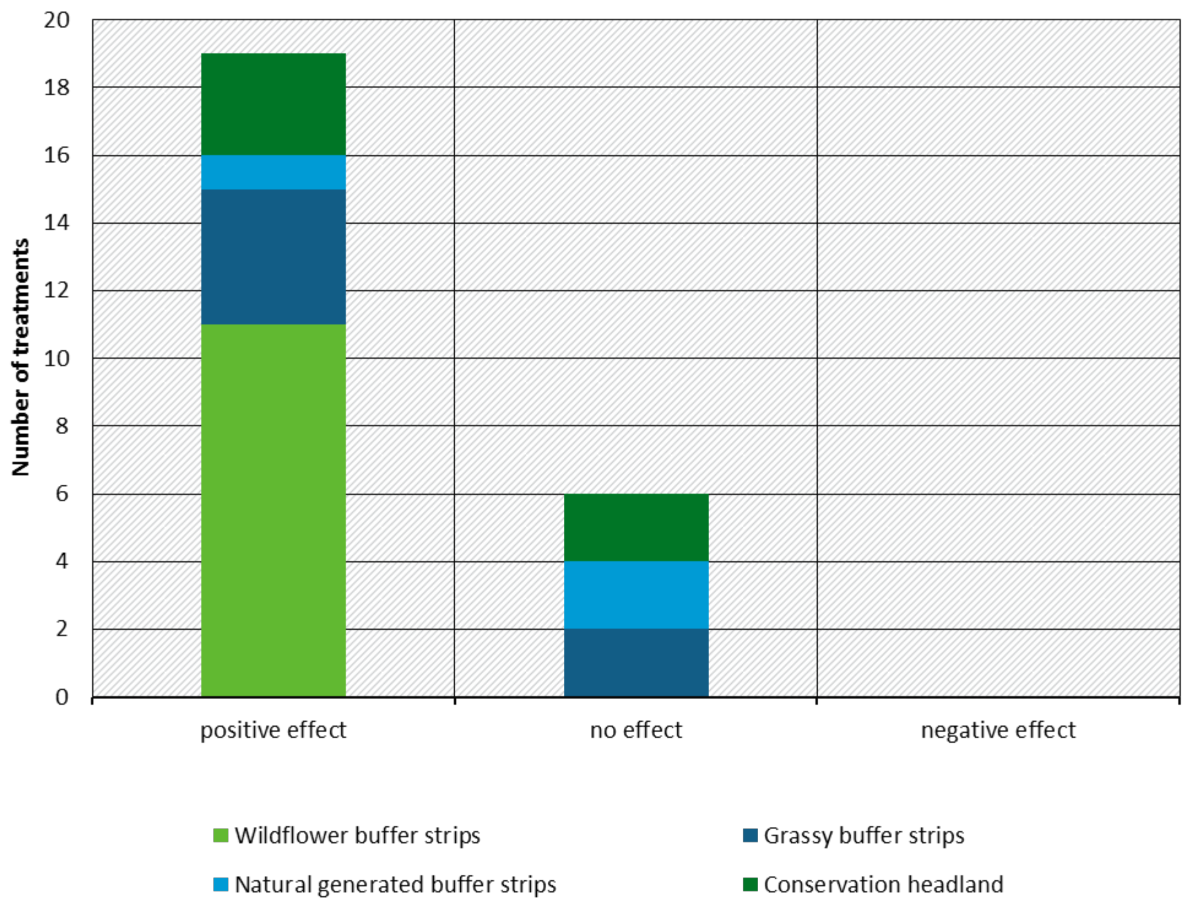
Table 43 and Figure 23 show that all studied in-field wildflower strips (or with nectar and pollen mixture) increased significantly abundance and in most cases also species richness of bees, butterflies and hoverflies (Meek et al. 2002; Pywell et al. 2005; Aviron et al. 2007; Carvell et al. 2007; Kohler et al. 2008; Haenke et al. 2009; Scheper et al. 2015; Wood et al. 2015; Oppermann et al. 2016). Conservation headlands, buffer strips with natural vegetation or grassed buffer strips seem to be less effective for bee populations (Table 43). Significant positive effects on bees of these types of in-field buffers could only be found in few cases, e.g. by Pywell et al. (2005) as a result of buffers with natural vegetation or by Marshall et al. (2006) as a result of implemented grassed buffer strips. In contrast, there are also studies which did not find a significant positive effect of conservations headlands (Carvell 2002; Pywell et al. 2006), buffer strips with natural vegetation (Carvell 2002; Meek et al. 2002; Carvell et al. 2004), or grassed buffer strips (Carvell 2002; Meek et al. 2002; Carvell et al. 2004) on bee abundance/richness. Also for butterflies the results with respect to conservation headlands and grass strips seem to be diverse. Significantly positive effects on butterflies by implementing conservation

[2] A treatment is defined as a different buffer strip type (of different width, if applicable) investigated in a study.

[3] The treatment 3-m wildflow-er/grass strips, hedge, 3-m grass strips (tussocky grass mixture) investigated by Meek et al. 2002 and Carvell et al. 2004 was excluded from this analysis because it is not possible to assign this treatment to one of the strip types (wildflower strip, grassy buffer strip, natural regenerated and conservation headland).

headlands and grassed buffer strips were shown by de Snoo et al. (1998), Field et al. (2005) and Field et al. (2007). In contrast, Meek et al. (2002) did not found a significant effect on butterfly abundance and species richness by implementing 6 m wide buffer strips sown with tussocky grass mixture and buffer strips with natural regenerated vegetation.

Figure 23: Effects of wildflower, grassy and natural regenerated in-field buffer strips as well as conservation headlands on FVI. Number of treatments demonstrating positive effects on FVI abundance and/or species richness, no effects or negative effects. Data derived from Table 43. Note, that the number of treatments do not correspond to the number of studies, because some authors investigated different treatments of in-field buffers in their study.



Source: own illustration, ecotox consult

Table 43: Effects of different in-field buffer strip types on flower-visiting insects (FVI) as investigated in field studies (n.r. = not reported). "Yes" means that a statistically significant effect was observed, unless otherwise is stated.

Reference	Buffer strip type	Buffer width/length	Positive effect on FVI		Investigated taxa	Adjacent crop	Study period
			abundance	species richness			
(Aviron et al. 2007)	Wildflower strips	n.r. / n.r.	**Yes**	**Yes**	Butterflies	dominantly wheat	5 years (sampling of butterflies every 2nd year)
(Carvell et al. 2007)	Wildflower strips	6 m / 50 m	**Yes**	**Yes**	Bumble bees	cereals	3 years
(Carvell et al. 2007)	Strip with nectar & pollen mixture	6 m / 50 m	**Yes**	**Yes**	Bumble bees	cereals	3 years
(Carvell et al. 2007)	Conservation headlands	6 m / 50 m	No	No	Bumble bees	cereals	3 years
(Carvell et al. 2007)	Natural regeneration	6 m / 50 m	No	No	Bumble bees	cereals	3 years
(Carvell et al. 2007)	Grass strips (tussocky grass mixture)	6 m / 50 m	No	No	Bumble bees	cereals	3 years
(de Snoo et al. 1998)	Conservation headlands	3 m / 100 m 6 m / 100 m [b]	**Yes**	**Yes**	Butterflies	winter wheat	2 years
(de Snoo et al. 1998)	Conservation headlands	3 m / 100 m 6 m / 100 m [b]	**Yes**[c]	**Yes**[c]	Butterflies	potatoes	2 years
(de Snoo 1999)	Conservation headlands	3 m / 100 m 6 m / 100 m [b]	**Yes**	n.r.	Hoverflies, Aphid predator (e.g. ladybirds)	winter wheat	2 years
(Field et al. 2005)	Grass strips	6 m / n.s.	**Yes**[a]	No	Butterflies	cereals	4 years
(Field et al. 2007)	Grass strips (tussocky grass mixture)	2 m / n.r.	**Yes**	n.r.	Butterflies	Mainly wheat, barley, field beans	4 years

Reference	Buffer strip type	Buffer width/length	Positive effect on FVI abundance	Positive effect on FVI species richness	Investigated taxa	Adjacent crop	Study period
(Kohler et al. 2008)	Wildflower strips	10 m / 10 m	**Yes**	**Yes**	Bees, Hoverflies	crop monocultures	1 year
(Haenke et al. 2009)	Grass strips	3 m wide	**Yes**	**Yes**	Hoverflies	wheat	1 year
(Haenke et al. 2009)	Flower strips	3-6 m wide	**Yes**	**Yes**	Hoverflies	wheat	1 year
(Haenke et al. 2009)	Flower strips	12-25 wide	**Yes**	**Yes**	Hoverflies	wheat	1 year
(Marshall et al. 2006)	Grass strips (3-years old)	6 m / n.r.	**Yes**	**Yes**	Bees	arable fields	1 year
(Meek et al. 2002)	Grass strips (tussocky grass mixture)	6 m / 72 m	No	No	Butterflies	cereals	1 year
(Meek et al. 2002)	Wildflower/grass strips	6 m / 72 m	**Yes**[d]	No	Butterflies	cereals	1 year
(Meek et al. 2002)	3-m wildflower/grass strips, hedge, 3-m grass strips (tussocky grass mixture)	6 m / 72 m	**Yes**[d]	No	Butterflies	cereals	1 year
(Meek et al. 2002)	Strips with natural regenerated	6 m / 72 m	No	No	Butterflies	cereals	1 year
(Meek et al. 2002) (Carvell et al. 2004)	Grass strips (tussocky grass mixture)	6 m / 72 m	No	n.r.	Bumble bees	cereals	3 years
(Meek et al. 2002; Carvell et al. 2004)	Wildflower/grass strips	6 m / 72 m	**Yes**	n.r.	Bumble bees	cereals	3 years

Reference	Buffer strip type	Buffer width/length	Positive effect on FVI		Investigated taxa	Adjacent crop	Study period
			abundance	species richness			
(Meek et al. 2002; Carvell et al. 2004)	3-m wildflower/grass strips, hedge, 3-m grass strips (tussocky grass mixture)	6 m / 72 m	**Yes**	n.r.	Bumble bees	cereals	3 years
(Meek et al. 2002; Carvell et al. 2004)	Natural regeneration	6 m / 72 m	No	n.r.	Bumble bees	cereals	3 years
(Oppermann et al. 2016)	Wildflower strips (annual & perennial)	Not specified. In total 10% of the 50 ha investigation area.	**Yes** [e]	**Yes** [e]	Wild bees	arable fields	4 years (additionally control survey 1 year before)
(Pywell et al. 2005)	Conservation headland	at least 6 m [f] / n.r.	No	No	Bumble bees	arable fields	1 year
(Pywell et al. 2005)	Natural regeneration	at least 6 m [f] / n.r.	**Yes**	**Yes**	Bumble bees	arable fields	1 year
(Pywell et al. 2005)	Wildflower strips	at least 6 m [f] / n.r.	**Yes**	**Yes**	Bumble bees	arable fields	1 year
(Scheper et al. 2015)	Wildflower strips	3 m / 100 m	**Yes**	**Yes**	Bumble bees; Solitary bees	arable fields	2 years (additionally control survey 1 year before)
(Wood et al. 2015)	Flower rich agri-environment schemes [g]	Not specified.	**Yes (also on bumble bee nest density)**	n.r.	Bumble bees	Mainly wheat, barley, oilseed rape, permanent/silage grassland	2 years

[a] Only the abundance of butterfly species *Maniola jurtina* and in consequence total butterfly abundance increased significantly on the 6 m grass margins.

[b] Data of 3 m wide and 6 m wide strips were pooled, as there was no significant difference between the groups.

[c] A significant difference was only observed in one of the two study seasons.

[d] Only the abundance of butterfly species *Maniola jurtina* and *Aphantopus hyperantus* increased significantly, and thus total butterfly abundance increased significantly.

[e] No statistical data analysis was performed.

[f] Field margins were established in the context of the Arable Stewardship Pilot Scheme. This requires a minimum margin width of 6 m.

[g] Flower rich environment schemes include pollen and nectar mixes on rotational strips/fields, floristically enhanced grass flower strips and maintenance, restoration, creation of species rich grassland.

Table 44 shows the number of findings of positive effects on wild bee, butterfly or hoverfly abundance and/or species richness, no effects or negative effects in available studies per each buffer strip type. Of all buffer strip types, wild flower strips are the best investigated buffer strip type, showing positive effects on investigated FVIs in all studies.

Table 44: Number of findings of positive (+), no (○), or negative (-) effects on wild bee, butterfly and hoverfly abundance and/or species richness. Furthermore results are shown in percent of total available findings per buffer strip type and FVI group. Data derived from Table 43. Note, that the number of findings do not correspond to the number of studies, because some authors investigated the effects of in-field buffer strips on several FVI taxa groups.

Type of in-field buffer strip	Wild Bees			Butterflies			Hoverflies		
	+	O	-	+	O	-	+	O	-
Wildflower strip	8	0	0	2	0	0	3	0	0
	100%	0%	0%	100%	0%	0%	100%	0%	0%
Grassy buffer strip	1	2	0	2	1	0	1	0	0
	33%	67%	0%	67%	33%	0%	100%	0%	0%
Natural regenerated	1	2	0	0	1	0	0	0	0
	33%	67%	0%	0%	100%	0%	0%	0%	0%
Conservation headland	0	2	0	2	0	0	1	0	0
	0%	100%	0%	100%	0%	0%	100%	0%	0%

The treatment 3-m wildflower/grass strips, hedge, 3-m grass strips (tussocky grass mixture) investigated by Meek et al. 2002 and Carvell et al. 2004 was excluded in total number because it is not possible to assign this treatment to one of the strip types (wildflower strip, grassy buffer strip, natural regenerated and conservation headland).

The number of forage flowers is important for the effectiveness of flower strips. This was shown for example by Carvell et al. (2007) who found a positive significant correlation between number of forage flowers and the abundance of bumble bees. Also Kohler et al. (2008) found in their study that diversity and abundance of hoverflies were related to flower abundance. Moreover, a diverse plant community present on buffer strips can favour abundance and diversity of FVI as shown by Gill et al. (2014) or Scheper et al. (2015) for bees. Consequently, the effectiveness of wildflower strips can be enhanced by increasing the amount of flowering plant species in the used seed mixture (Scheper et al. 2015). Moreover, the adding of perennial plant species in seed mixtures can further improve buffers as habitats for FVI (Gill et al. 2014). Also Oppermann et al. (2013) and Schmid-Egger & Witt (2014) underline the importance of perennial flower strips. Advantages of perennial strips are for example the availability of habitats also in winter and there is generally more time available for species to colonize the buffer strip (Fenchel et al. 2015).

To promote the presence of food specialized FVI species the seed mixtures should ideally contain the specific host plant species of the FVI species of concern (Gill et al. 2014). Backman & Tiainen (2002) studied the abundance of bumble bees in field margins in a Finnish farmland area. They found the availability of certain key plant species in field margins to be the most important factor to attract bumble bees. The importance of the presence of key plant species is also underlined by results of other studies (Meek et al. 2002; Carvell et al. 2007; Cole et al. 2015) which found that the majority of plant-pollinator interactions occurred on just a few plant species. For an effective management, it is furthermore important that suitable flowers for pollinators are present during the whole summer.

Numerus authors (Oppermann et al. 2013; Schmid-Egger & Witt 2014; Fenchel et al. 2015) highlight the fact that it is important that sown wildflower seed mixtures contain indigenous plant species.

Thus, regional site characteristics can be considered. The use of regional wildflower seed mixtures is mandatory in Germany (§ 40 (4) BNatSchG) from the year 2020 (Schmid-Egger & Witt 2014).

Broader field margins and a high density of flowering margins in the agricultural landscapes promoted the abundance of bumble bees in the study of Backman & Tiainen (2002). Also Cole et al. (2015) detected a positive influence of buffer strip width[4] on pollinator abundance and diversity. Butterflies were more abundant and more diverse in wide buffer strips (width > 5 m) in comparison with narrow buffer strips (width ≤ 3.5 m). Also bumble bees were found more abundant in wider buffer strips. In contrast, there were no differences between 3 and 6 m wide buffer strips regarding butterflies and hoverflies in the study of de Snoo (1999) and de Snoo et al. (1998).

The efficiency of in-field buffer strips to support FVI populations might be reduced, when there are still residues of systemic pesticides on implementation areas resulting from former agricultural use. These residues might be taken up by roots of the wildflower plants, and in consequence be toxic for FVIs feeding on this plant. That wildflower plants on field margins can contain neonicotinoid (which belong to the class of systemic pesticides) residues was for example shown by Botias et al. (2015). Therefore, at best, in-field buffer strip should be implemented when a dissipation of former used systemic pesticides can be assumed.

8.2.1.3 Feasibility and acceptability by farmers

In-field buffer strips might be easily implemented as risk mitigation measures (Reichenberger et al. 2007; Bereswill et al. 2014). However, farmers have additional effort for the implementation and maintenance (e.g. seeding, cutting, ploughing). Moreover, there will be slight yield losses because a certain amount of field area is taken out of production. Therefore, the implementation of in-field buffers is supposed to be accepted only in case that there are possibilities for financial compensation (see also section 8.5).

Moreover farmers might be concerned about higher weed abundance in crop in consequence of implemented in-field buffer strips (Berger & Pfeffer 2011). In an experimental study of Balzan et al. (2016) wildflower strips lead to increased Lepidopteran-caused pest damage in adjacent tomato fields. Nevertheless, tomato production was higher in these fields adjacent to wildflower strips. Increased crop productivity might be caused by higher pollination due to higher abundance of pollinating insects. Several authors (Marshall et al. 2006; Feltham et al. 2015) showed that in fields with adjacent wildflower/grass buffer strips pollinating insects are more abundant than in fields without such strips. The higher the abundance of pollinating insects in crop, the higher might be the pollination and crop yield might increase. For example, Garibaldi et al. (2013) found for 41 crops higher crop yields with increasing visits by pollinators. Such higher crop productivity might positively influence the acceptability of in-field buffer strips by farmers.

According to such a calculation 5 m wide buffers adjacent to field crops might provide a 79% pesticide spray drift reduction in comparison to the pesticide deposition which is expected at a distance of 1 m. In case of adjacent fruit crops and vineyards, respectively, 5 m wide buffers might provide 32-47% and 55% pesticide spray drift reduction in comparison to the pesticide deposition expected at a distance of 3 m; higher efficacies of 60-77% and 85% can be reached by 10 m wide buffers adjacent to fruit crops and vineyards, respectively

[4] Cole et al. (2015) investigated riparian buffer strips, however, their results might also be valid for buffer strips adjacent to terrestrial habitats.

In short: in-field buffer strips

- Pesticide spray drift reduction efficiency is dependent on the width of in-field buffer strips and decreases exponentially with increasing distance. Efficient reduction can be provided by 5 m wide buffers adjacent to field crops (79% drift reduction), and 10 m wide buffers adjacent to fruit crops (60-77% drift reduction) and vineyards (85% drift reduction).
- In-field wildflower strips (or with nectar and pollen mixture) are most effective in increasing abundance and species richness of bees, butterflies and hoverflies. Conservation headlands, buffer strips with natural vegetation or grassed buffer strips seem to be partly less effective for bee and butterfly populations.
- The number of forage flowers is important for the effectiveness of flower strips.
- Seed mixtures should contain key plant species of the FVI species of concern to promote the presence of food specialized FVI species.
- Regional wildflower seed mixtures containing indigenous plant species should be used.
- In general, FVI abundance might increase with increasing buffer width, because of higher amounts of food and nesting possibilities.
- In-field buffer strips might be easily implemented as risk mitigation measures. Acceptability by farmers might be increased financial compensation possibilities.

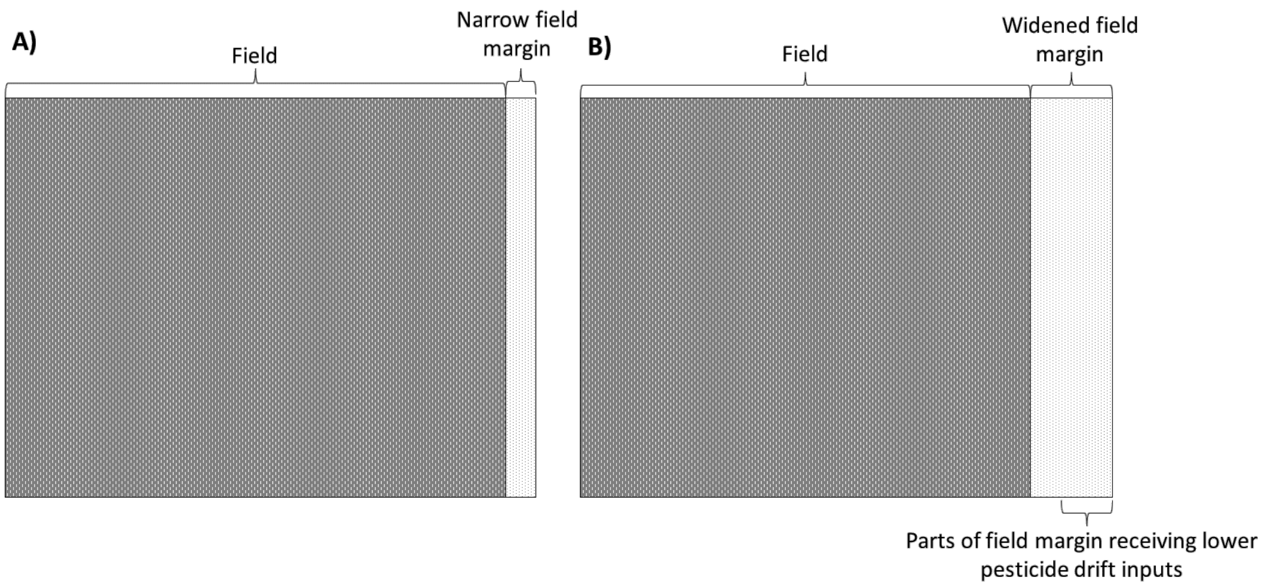
8.2.2 Extension of small field margins

8.2.2.1 Reduction of pesticide entries

Field margins are permanently present, usually linear habitats adjacent to agricultural fields with grassy, herbaceous or higher vegetation (such as hedges, shrubs, trees). Narrow field margins with widths around 1-2 m are commonly present in intensively used agricultural regions as shown by Hahn et al. (2014) for the agricultural landscape in Rhineland-Palatinate (Germany). The extension of these small field margins resulting in a reduction of pesticide deposition in off-field habitats because remote parts of the margin receive lower pesticide spray drift entries (Brühl et al. 2013) as a result of increasing the distance between these parts of the margin and the field (Figure 24).

Moreover, wider permanent field margins offer the opportunity for the establishments of higher vegetation such as shrubberies or hedges, which might further promote the reduction of spray drift entries in off-field habitats (see section 8.2.4.1). For example, Ohliger & Schulz (2010) observed in a landscape monitoring conducted in the vineyard area Palatinate (Germany) that a minimum margin width of 6 m was necessary for the presence of high vegetation.

Figure 24: Extension of small field margins. The extension of the narrow field margin (A) results in lower pesticide spray drift entries in remote parts of the widened margin (B).



Source: own illustration, ecotox consult

8.2.2.2 Effects on FVI

Because some parts of the margin receive lower pesticide spray drift entries also the pesticide exposure of FVI present in this area is reduced. Furthermore, the widening of available permanent field margins should generally increase habitat availability for FVIs and thus also food resources, nesting possibilities and refuges. This assumption is supported by studies of Backman & Tiainen (2002) in an agricultural landscape in Southern Finland: field margin width was significantly positively related to bumble bee density. Also Cole et al. (2015) found a higher pollinator abundance (butterflies and bees) and higher butterfly abundance in wide riparian buffer strips (width > 5 m) than in narrow buffer strips (width ≤ 3.5 m). This relationship might also be valid for buffer strips adjacent to terrestrial habitats. Rands & Withney (2011) demonstrated using a model that increasing field margin width should lead to an increase of forage availability for bees. Based on their model data, the authors concluded that wider field margins might be beneficial for long-distance foragers (e.g. honeybees) as well as short-distance foragers (e.g. solitary bees). The only exceptions might be bees with a very short forage range of < 125 m.

However, the efficacy of this mitigation measure is highly dependent on the specific field margin characteristics. With respect to the protection of FVI, it seems to be important to manage field margins for example regarding the presence of flowering plants, which were shown as very important to benefit FVI populations (Carvell et al. 2007; Kohler et al. 2008; Scheper et al. 2015).

The widening of field margins might be of special importance in the agricultural landscape because in contrast to in-field buffer strips, field margins are permanently present. Thus, the plant species composition present on these margins differs from in-field buffer strips which are ploughed in certain rhythms. For example Kuussaari et al. (2011) observed in field margins higher species richness of Lepidoptera than in adjacent fallows which is assumed by the authors to be a result of missing butterfly larval host plant species in the fallows. Moreover, plant structures such as hedges and shrubberies which also favour some FVI species (see section 8.2.4) can only be present on permanent field margins of sufficient size.

8.2.2.3 Feasibility and acceptability by farmers

The widening of present field margins is commonly implemented in Germany in the context of land-consolidation arrangements ("Flurbereinigungsverfahren"). In these cases agricultural land which is required for the widening of field margins is bought by communes, which consequently implement this measure. Feasibility might therefore be difficult (Bereswill et al. (2014)), at least in regions with highly fertile soils.

As already outlined in Chapter 8.1, use restrictions determined in the German pesticide authorization process for specific pesticide uses are only mandatory when applying a pesticide neighbouring a terrestrial off–field habitat with a width of 3 m or more. Therefore, the extension of narrow field margins to width of 3 m and more means for the farmers, that for the application of certain pesticides use restrictions (such as no-spray zones) might become mandatory. Thus, acceptability of this measure by farmers might be rather low. However, via the creation of wider field margins also the percentage of small structures in the agricultural area might increase, which might result in the registering of the commune in the German "Kleinstrukturenverzeichnis". When listed in this register, many pesticide use restrictions for the protection of terrestrial off-field habitats (e.g. distance requirements or use of drift reducing techniques) have not to be followed in this region. This might consequently increase also acceptability of this measure.

In short: extension of small field margins

- The extension of small field margins results in a reduction of pesticide deposition in off-field habitats, because remote parts of the margin receive lower pesticide spray drift entries as a result of increasing the distance between these parts of the margin and the field.
- With increasing width of field margins FVI abundance should increase, because of higher amounts of habitat, food and nesting possibilities.
- The efficacy of this measure on FVI populations is highly dependent on the specific field margin characteristics (e.g. present flowering plants) and should be managed appropriately.
- The widening of field margins might be of special importance in the agricultural landscape because these habitats are permanently present for FVI.
- Feasibility might be difficult at least in regions with highly fertile soil, and acceptability by farmers is supposed to be rather low.

8.2.3 Creation of conservation fallows

8.2.3.1 Reduction of pesticide entries

Land lying fallows (arable fallow) are defined as agricultural fields which are temporarily taken out of production (Toivonen et al. 2016) and where consequently pesticides are not applied. In general, arable fallows can be left to the natural regeneration process or wildflower mixtures can be seeded (BMEL 2015b).

By implementing arable fallows, the total amount of pesticides applied in the agricultural area is reduced (because on fallows application of pesticides is generally not allowed). Furthermore, depending on width, length and location, arable fallows can also lead to a decrease of pesticide spray drift entries in off-field habitats. For example, if a fallow is located between a cultivated field and a field margin, the distance between pesticide application (on the cultivated field) and the off-field habitat (i.e. field margin) is increased. With increasing distance pesticide spray drift entries exponentially decrease (Ganzelmeier et al. 1995; Rautmann et al. 2001; Van de Zande et al. 2015).

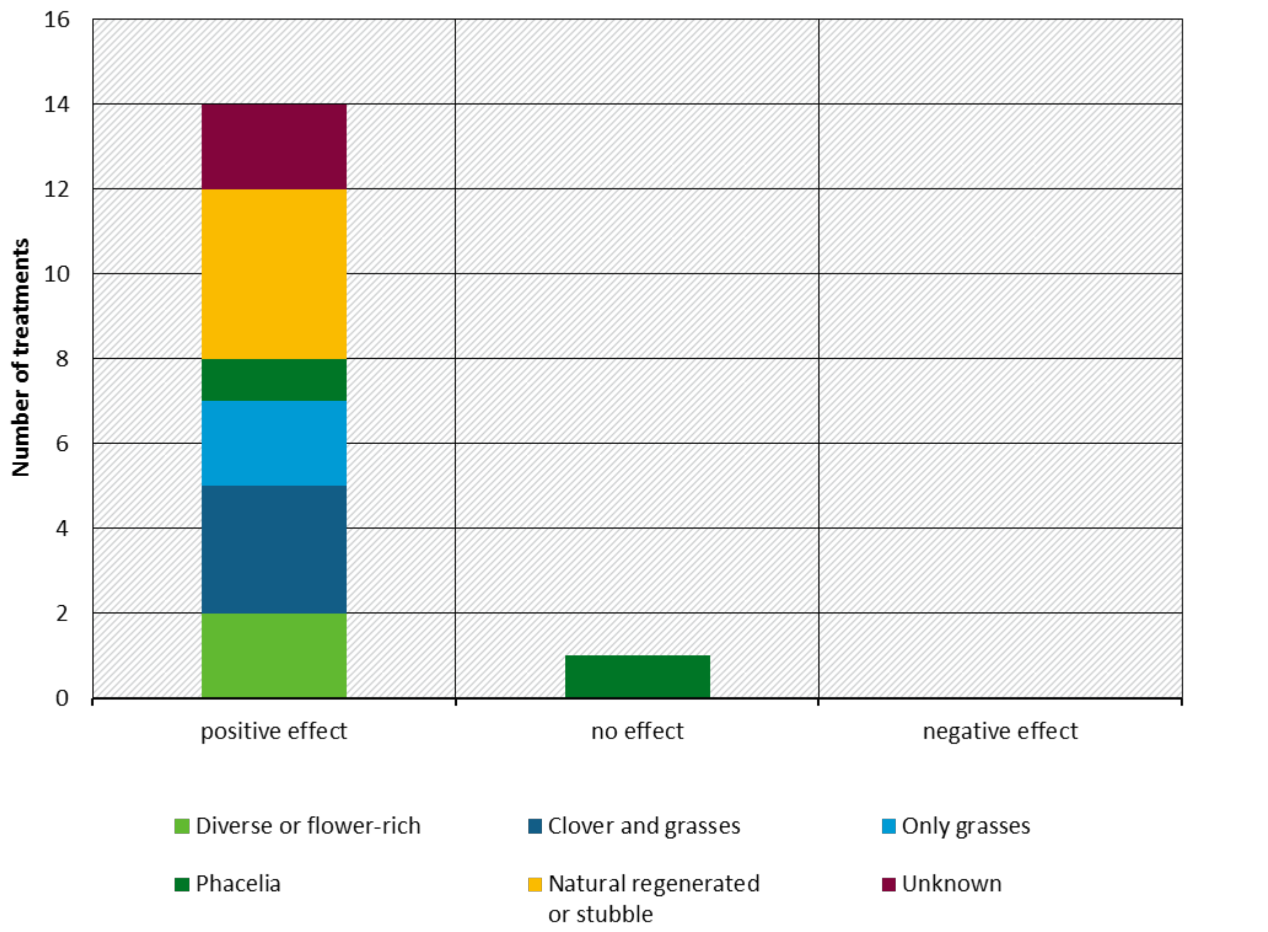
8.2.3.2 Effects on FVI

Fallows can promote FVI populations by reducing the pesticide exposure of FVI in non-target habitats and by enhancing habitat availability. There are numerous studies in literature available investigating the effects of different fallow types on FVI populations. The main results of these studies are presented in Table 45 and are shortly described in the following.

The results of reviewed literature indicate that land lying fallows might positively affect species richness and abundance of FVI (Table 45). Nearly all investigated fallow types had a positive effect at least on FVI abundance or species richness (see also Figure 25). Only in one case no positive effect (neither on abundance nor species richness) could be found: in fallows sown with *Phacelia* plants (Gathmann et al. 1994) on wild bees and wasps.

Positive effects of fallows on bumble bees and other wild bees were reported by numerous studies (Gathmann et al. 1994; Steffan-Dewenter & Tscharntke 2001; Diekotter et al. 2006; Alanen et al. 2011; Kuussaari et al. 2011; Holland et al. 2015). Further positive effects were observed on Lepidoptera, hoverflies and wasps by Denys & Tscharntke (2002), Kohler et al. (2008), Alanen et al. (2011), Kuussaari et al. (2011) and Holland et al. (2015).

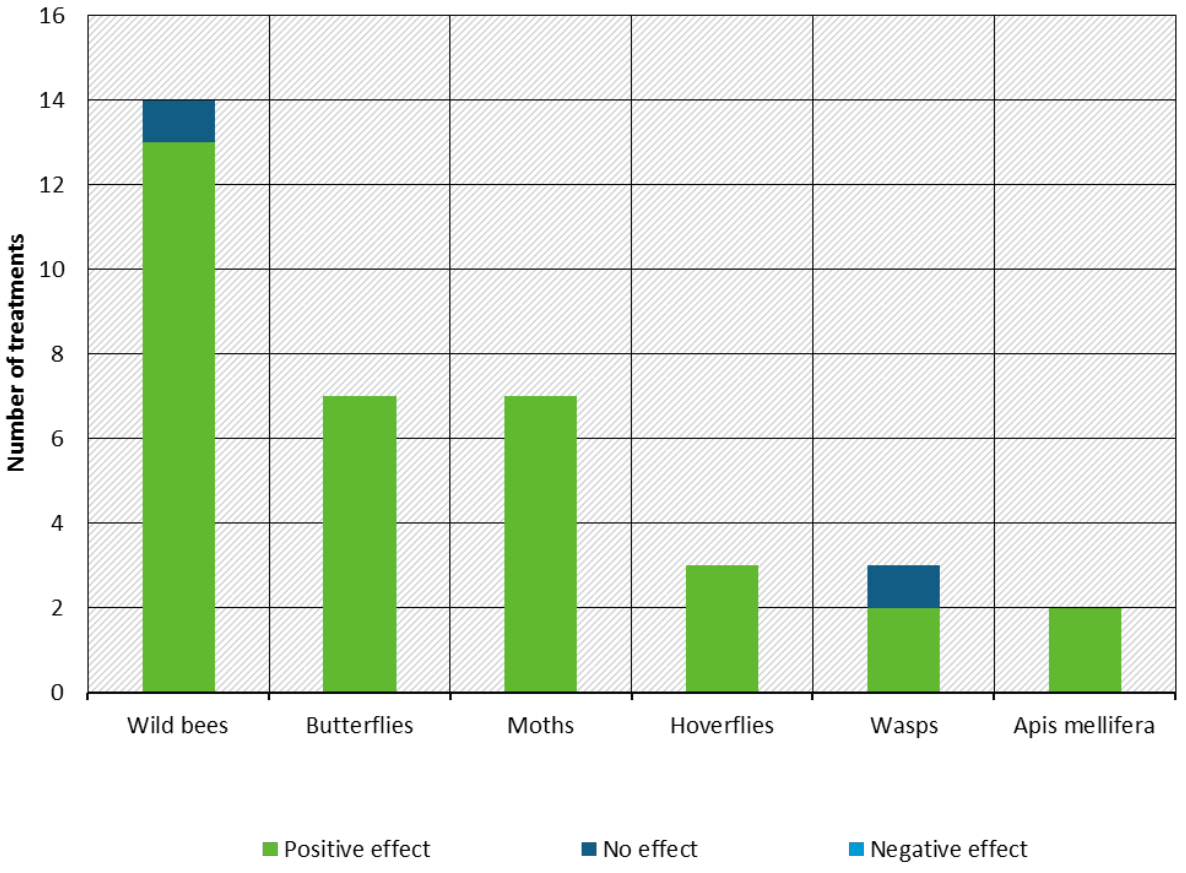
Figure 25: Effects of fallows on FVIs (bees, butterflies, hoverflies, moths, wasps). The number of available treatments, demonstrating the effect of different fallow types on FVI abundance and/or species richness, is shown. Note, that the number of treatments do not correspond to the number of studies, because some authors investigated different treatments of fallows in their study.



Source: own illustration, ecotox consult

Figure 26 shows that wild bees are the most investigated taxa (see also Table 45). But also for other FVI taxa such as butterflies, moths, hoverflies, wasps and *Apis mellifera* are data available. Positive effects of fallows on FVI abundance and/or species richness were shown for all investigated FVI groups.

Figure 26: The number of fallow treatments investigated in studies found in literature (Table 45), demonstrating a positive effects on FVI (bees, butterflies, moths, hoverflies and wasps) abundance and/or species richness, no effect or negative effect. Data is derived from Table 45. Note, that the number of findings do not correspond to the number of studies, because some authors investigated the effects of fallows on several FVI taxa.



Source: own illustration, ecotox consult

As outlined for in-field buffer strips (please refer to section 8.2) the abundance of forage flowers is important for the effectiveness of land lying fallow. For example Kohler et al. (2008) reported a positive influence of flower abundance on FVI populations. In addition, a high degree of plant diversity on fallows is another (maybe even more) important characteristic. So Kuussaari et al. (2011) found that flowering species richness had positive significant effects on FVI abundance while flower coverage had not. Therefore seed mixtures for fallows should provide high plant diversity and flower coverage. Especially fallows sown with seeds of *Agrostis capillaris, Festuca ovina* and special nectar and pollen mixtures highly attracted bumble bee species (Alanen et al. 2011; Kuussaari et al. 2011). Furthermore, *Trifolium pratense, Phacelia tanacetifolia* (Steffan-Dewenter & Tscharntke 2001; Carreck & Williams 2002; Diekotter et al. 2006) and *Raphanus sativus* (Kohler et al. 2008) might provide a positive contribution to the occurrence of bumble bees. In addition, Diekotter et al. (2006) found a high density of *Bombus muscorum* nests near *T. pratense.* Furthermore, bumble bees can also be attracted by *Phacelia* plants (Pontin et al. 2006). However, Gathmann et al. (1994) showed that fallows only sown with *Phacelia* seeds might be insufficient since FVI species richness is highly dependent on the flower diversity. The sowing of long-term seed mixtures (perennial plants like *T. pratense, A. capillaris* and *F. ovina*) might be a habitat enhancing method to increase biodiversity of FVIs (Nitsch et al. 2016). In this context, Steffan-Dewenter & Tscharntke (2001) found that high number of uncommon rare bumble bees species (e.g. *Andrena distinguenda* and *Andrena lagopus)* were more attracted to natural regenerated fallows than to fallows sown with *P. tanacetifolia* (sown annual plant). Thus, diverse and attractive plant communities sown on fallows as well as a sufficient supply of nectar and pollen seem to be an important factor for the occurrence of FVIs, particularly bumble bees.

Fallows might have generally less positive influences on the occurrence of Lepidoptera species (butterflies, moths) compared to wild bees (Alanen et al. 2011; Kuussaari et al. 2011). Alanen et al. (2011) and Kuussaari et al. (2011) observed, that bumble bees highly preferred to visit fallows, while Lepidoptera were more attracted by surrounding field margins. Lepidoptera species have other foraging strategies than wild bees (Kuussaari et al. 2011) and are strongly dependent on larval host plants as outlined by Holland et al. (2015). For example the butterfly species *Pseudophilotes baton* is specialized on its larval host plants *Thymus pulegioides* and *T. serpyllum* (for further information, please refer to section 0). This might be the reason why butterflies and moths preferred to visit surrounding field margins with high densities of larval host plants instead of established fallows (Alanen et al. 2011; Kuussaari et al. 2011). Kuussaari et al. (2011) found in their study that 38% of all observed Lepidoptera did not even visit the fallows.

Perennial fallows provide perennial habitats with additional possibilities of early foraging in spring-time (Nitsch et al. 2016); thus FVIs can forage at most time of the year. In general, butterflies are mainly active in the early-season (mid-May to mid-June) and bumble bees occurred rather in the early to mid-season (mid-May to early August) (Holland et al. 2015). For different flight durations of bees please refer to Figure 11. Throughout the year bee species flight activity varies considerably (Figure 10). As early as February some species begin to actively forage. From March to September at least 25% of bee species are simultaneously active, from May to July around 70%. June is the month with the highest number of flying species (76%) (data from Roberts et al. (unpublished)).

The age of fallows might have an additional influence on diversity and abundance of FVI communities. Several studies are showing positive effects of successional age of fallows on FVIs (Gathmann et al. 1994; Kuussaari et al. 2011; Toivonen et al. 2015). However, there are also several studies showing no fallow age dependent effects (Steffan-Dewenter & Tscharntke 2001; Denys & Tscharntke 2002; Alanen et al. 2011). For example Kuussaari et al. (2011) showed that 2-year fallows had significantly higher effects on FVI abundance and species richness compared to 1-year fallows. However, Alanen et al. (2011) highlighted that bumble bee abundance was not significantly influenced by the age of the fallow. In Toivonen et al. (2015) a higher abundance of bumble bees in short-term fallows (3-4 years) and a higher abundance and species richness of butterflies in long-term fallows (>8 years) were observed. In contrast, Toivonen et al. (2016) found more butterflies in 3-4 year fallows than in fallows with an age of more than 8-years while bumble bees had no specific preference. Steffan-Dewenter & Tscharntke (2001) detected no significant effect of the successional age of the fallow on wild bees. In contrast, using a similar study design Gathmann et al. (1994) found an influence of successional age of fallows on wild bee and wasp species richness. Based on these studies, there is no clear evidence of generally positive effects on FVIs by long-term fallows.

Differences in species specific responses of FVIs to different old fallows could be caused by a more rapid adaption of wild bees to newly created habitats compared to other FVIs (butterflies, moths), since wild bees tend to occur already in early years, whereas e.g. Lepidoptera mainly occurred in later years (Alanen et al. 2011; Kuussaari et al. 2011). For this reason, it seems to be advisable to establish fallows at least for several years to provide good habitat conditions for a wide range of FVI species.

Higher percentage of fallow lands adjacent to crops may increase FVI abundance and species richness as shown by Holland et al. (2015). Only hoverflies were not influenced by increasing the percentage of uncropped fallow land. The authors assumed that this could be due to the greatly high mobility range of hoverflies. However, also the insufficient supply of attractive resources for hoverflies in agricultural landscapes could be a reason for less influence on hoverfly abundance and species richness (Holland et al. 2015). Denys & Tscharntke (2002) found an influence of large uncropped areas since significant more moth species were found in large fallows (>1 ha) than in narrow field margins (3m wide, 100-150 m long).

Communities of FVI on fallows are further influenced by the landscape type of the surrounding area. Toivonen et al. (2016) and Toivonen et al. (2015) observed in Finish landscapes that butterfly species were more abundant in fallows surrounded by forest while bumble bees had no special needs in surrounding landscapes (they were attracted by both, fallows surrounded by forested and opened landscape). Toivonen et al. (2016) recommended a heterogeneity and variety in fallows which might provide comprehensive habitat types in order to increase abundance of single FVI species and diversity of FVI communities. Furthermore, there are some indications that semi-natural nature reserves[5] present in the neighbourhood of established fallows may have a positive influence on FVI abundance and species richness as demonstrated by Kohler et al. (2008) for hoverflies. With decreasing distance to nature reserves, flower abundance as well as the total species density and abundance of hoverflies rapidly increased in fallow patches. However, this relationship was not observed for bee species.

Flower-visiting insect species richness in naturally generated fallows might additionally benefit from mowing of perennial fallows in early season (once per year). Excessive growth of annual plant species can be suppressed by an annual mowing interval, resulting in a higher plant species richness, which might support a high species richness of FVI communities in fallows (Gathmann et al. 1994). However, Alanen et al. (2011) found either positive or negative significant effects of mowing activities (mown with a mower chopper between late summer and autumn) on compositions of FVI communities. In order to create a diverse fallow structure Gottwald & Stein-Bachinger (2016) recommended reduced mowing activities (only on sub-areas) for promoting FVI foraging resources and nesting habitats. This is in agreement to the recommended reduced mowing activities of off-filed habitats such as field margins or meadows as discussed in section 8.2.10.

8.2.3.3 Feasibility / acceptance

In 2002 Jacot et al. interviewed farmers about their experiences with fallows (established after 1995 and subdivided in 3 year and 6 year fallows). The main criteria of the survey were effort, development, type of establishment and potential problems of fallow maintenance. Seventy-five percent of all interviewed farmers assessed the effort of implementation of general fallows as relatively low. In general, after the establishment of fallows no extensive conversation measure is needed (only regular monitoring and occasional mowing). However, farmers mentioned the growth of unintentional weeds (also indicated by (Sattler & Nagel 2010)) because of the official restrictions not to remove them with herbicides as a main maintenance problem. The growth of grassy plants is rising with the age of the fallow (from third year) but it was assessed as harmless by farmers. According the survey of Jacot et al. (2002) 70% to 80% of the Swiss population (farmer and non-agricultural people) supported the establishment of flower diverse fallows. Also the image of farmers would benefit by the implementation of fallows.

Based on these results it can be assumed that the establishment of fallows is easy feasibly and also the acceptability by farmers might be moderate to high. However, the acceptability of fallows by farmers might be highly dependent on the productivity of the agricultural land. At sites with low agricultural

[5] The authors investigated Dutch semi-natural reserves (size of few to a few tens of hectares) consisting of mosaics of grassland, heathland, woodlots and occasional pools.

production farmers might be more poised to take land out of production (Sattler & Nagel 2010; Nitsch et al. 2016).

In short: land lying fallow

- Depending on width, length and location of the fallow, pesticide spray drift entries in off-field habitats might be reduced.
- Land lying fallow might positively affect FVI abundance and/or species richness. Positive effects were shown for bumble bees, wild bees, wasps, butterflies and moths.
- The flower diversity as well as the flowering coverage are important factors for FVI communities in terms of abundance and diversity. In addition to wildflower mixtures containing a multitude of pollen and nectar resources, also larval host plants should be sown to increase the attractiveness for diverse FVI communities.
- The Impact of fallows created by natural regeneration is not comprehensively investigated. However, there are indications that due to a high degree of diversity of plant communities natural regenerated fallows might be an effective tool to protect FVIs.
- Perennial fallows are providing foraging habitats in early season, because of a wider range of resource availability. Thus, FVI groups - which are already active during the early season – benefit from the supply of flowers in spring.
- The implementation of fallows might be easy feasible and acceptability by farmers might be moderate to high depending on the productivity of agricultural sites.

Table 45: Statistically significant effects of different fallow types on flower-visiting insects (FVI) as investigated in field studies (n.r. = not reported).

Reference	Fallow type	Fallow size	Positive effect on FVI		Investigated taxa	Adjacent crop	Monitoring period
			Abundance	Species richness			
(Alanen et al. 2011)	Undersown with diverse seed mixtures	0.25 ha	**Yes**	**Yes**	Bumble bees Butterflies Moths	Cereal	6 years
(Alanen et al. 2011)	Undersown with standard (competitive) mixtures	0.25 ha	No	**Yes**[a]	Bumble bees	Cereal	6 years
(Alanen et al. 2011)	Undersown with standard (competitive) mixtures	0.25 ha	**Yes**	**Yes**	Butterflies Moths	Cereal	6 years
(Alanen et al. 2011)	Undersown with less competitive mixtures	0.25 ha	No	**Yes**[a]	Bumble bees	Cereal	6 years
(Alanen et al. 2011)	Undersown with less competitive mixtures	0.25 ha	**Yes**	**Yes**	Butterflies Moths	Cereal	6 years
(Denys & Tscharntke 2002)	Naturally vegetated fallow	>1 ha	n.r.	**Yes**	Moths Hoverflies	Cereal	6 years
(Diekotter et al. 2006)	Fallow lands	n.r.	**Yes**	Not relevant	*B. muscorum*	Agricultural field	2 years
(Gathmann et al. 1994)	Sown fallows (Phacelia plants)	0.2–0.7 ha	n.r.	No	Wild Bees Wasps	Cereal	1 year
(Gathmann et al. 1994)	Sown fallows (clover-grass mixture)	0.2–0.7 ha	n.r.	**Yes**[b]	Wild Bees Wasps	Cereal	1 year

Reference	Fallow type	Fallow size	Positive effect on FVI Abundance	Positive effect on FVI Species richness	Investigated taxa	Adjacent crop	Monitoring period
(Gathmann et al. 1994)	Naturally vegetated fallows	0.2–0.7 ha	n.r.	**Yes**[b]	Wild Bees Wasps	Cereal	2 years
(Holland et al. 2015)	Fallow	1.5-6 ha	**Yes**	**Yes**	Wild bees Butterflies Hoverflies	Agricultural field	3 years
(Kohler et al. 2008)	Fallows	0.01 ha	**Yes**	**Yes**	Bees Hoverflies	Crop monoculture	1 year
(Kuussaari et al. 2011)	Undersown or normally sown with competitive seed mixtures	0.3 ha	n.r-	Not relevant	*Apis mellifera*	Barley	2 years
(Kuussaari et al. 2011)	Undersown or normally sown with competitive seed mixtures	0.3 ha	**Yes**[c]	**Yes**[c]	Bumble bees Butterflies Moths	Barley	2 years
(Kuussaari et al. 2011)	Stubble fallow	0.3 ha	**Yes**	Not relevant	*Apis mellifera*	Barley	1 year
(Kuussaari et al. 2011)	Stubble fallow	0.3 ha	**Yes**[c]	**Yes**[c]	Bumble bees Butterflies Moths	Barley	1 year
(Kuussaari et al. 2011)	Undersown or normally sown with less competitive seed mixtures	0.3 ha	**Yes**	Not relevant	*Apis mellifera*	Barley	2 years
(Kuussaari et al. 2011))	Undersown or normally sown with less competitive seed mixtures	0.3 ha	**Yes**[c]	**Yes**[c]	Bumble bees Butterflies Moths	Barley	2 years

Reference	Fallow type	Fallow size	Positive effect on FVI		Investigated taxa	Adjacent crop	Monitoring period
			Abundance	Species richness			
(Steffan-Dewenter & Tscharntke 2001)	Fallow sown with *Phacelia tanacetifolia*	0.6 ± 0.4 ha	**Yes**	No	Wild bees	Cereal	1 year
(Steffan-Dewenter & Tscharntke 2001)	Naturally vegetated fallows	0.6 ± 0.4 ha	**Yes**	**Yes**	Wild bees	Cereal	1-5 years

[a] Fallows undersown with standard (competitive) mixtures had a positive effect on FVI species abundance, however, fallows undersown with diverse seed mixtures were more effective.

[b] Also the nest number of bees and wasps were significantly higher.

[c] Species richness and abundance of moths and butterflies were lower in fallows compared to the field margins. But when comparing moths and butterflies with cereals as control they had higher species richness and abundance in fallows.

8.2.4 High vegetation such as hedges, shrubberies, trees

8.2.4.1 Reduction of pesticide entries in off-field habitats

The planting of high vegetation such as hedges, shrubberies and trees are commonly recommended for the reduction of pesticide spray drift entries in adjacent non-target areas (Candolfi et al. 2000; Reichenberger et al. 2007; Bereswill et al. 2014). The reduction of pesticide spray drift is caused by two main processes: the reduction of wind velocity and the interception of spray drift particles by the vegetation (Raupach et al. 2001). This results in a reduced spray drift deposition in the downwind area directly behind the windbreak (De Schampheleire et al. 2009). The characteristics of vegetation height and vegetation density highly influence the extent of pesticide reduction (Ucar & Hall 2001). For an effective reduction of entries via drift vegetation heights of 1.5 to 2 times higher than crop heights are recommended (Ucar & Hall 2001).

According to Bereswill et al. (2014) high riparian vegetation (shrubs and trees) in full leaf stage can provide a 75-95% (25^{th}-75^{th} percentile; median = 89%; n = 13) reduction efficiency. These values were derived based on the data of seven studies found in literature which investigated the spray drift reduction efficacy of high vegetation (Davis et al. 1994; Walklate 2001; Richardson et al. 2002; Wolf et al. 2004; Wenneker et al. 2005; Lazzaro et al. 2008; Vischetti et al. 2008). However, in spring when foliage is only sparse, spray drift reduction might be much lower. Bereswill et al. (2014) assume based on studies by Richardson et al. (2004), Wenneker et al. (2005) and Lazzaro et al. (2008) that spray drift reduction efficiency of vegetation with sparse foliage might be on average by a factor of 1.8 lower than reduction efficiency of vegetation in full leaf stage. Similar values were also proposed by the FOCUS working group on Landscape and Mitigation Factors in Ecological Risk Assessment (2007): they assume based on a literature survey a 25%, 50% and 75% spray drift reduction rate for high vegetation with a sparse foliage, in intermediate growth stage and in full leaf stage, respectively. Similar values for high riparian vegetation in full leaf stage were also reported by Schweizer et al. (2013) who found a 78-95% drift reduction provided by a hawthorn hedge (height: 4.4 m; width: 0.85 m; optical density: 82%) adjacent to an apple plantation. These reported reduction efficiencies for high riparian vegetation are also valid for high vegetation adjacent to terrestrial off-field habitats.

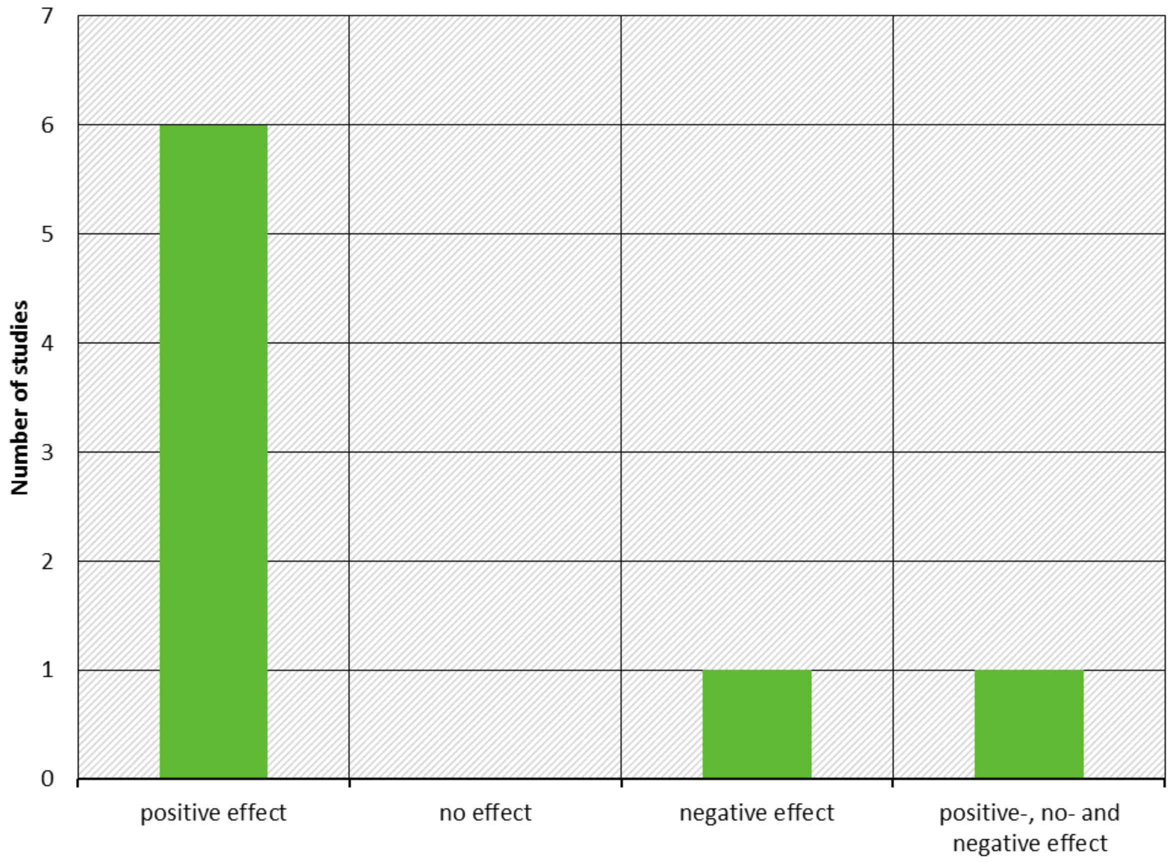
In case of tree rows or single trees a close understory is often lacking. Consequently a dense vegetation barrier over the entire vegetation height cannot be provided. In this case, spray drift reduction efficiencies are assumed to be lower, because the filter barrier is patchy.

It should be noted, that the above mentioned pesticide reduction efficacies are only valid for the area behind the hedge. The hedge itself will still receive pesticide spray drift entries (as for example shown by Koch et al. (2003)).

8.2.4.2 Effects on FVI

The presence of hedges, higher trees and other vegetated elements may enhance the habitat availability for FVI populations in agricultural landscapes. There are numerous studies in literature available investigating the effects of different hedges or trees on FVI populations (Table 46). In total eight studies were found (Figure 27). Six authors showed positive effects on FVI abundance and/or species richness, one author showed negative effects and one author showed positive-, no- and negative effects on different FVI taxa , respectively (Table 46, Figure 27). In the following the most important results of the studies are shortly described and analysed.

Figure 27: Number of treatments investigated in available literature studies (Table 45), demonstrating effects of high vegetation on FVIs (bees, butterflies, hoverflies, and moths). The number of available studies demonstrating the effect of hedges and trees on FVI abundance and/or species richness is shown.



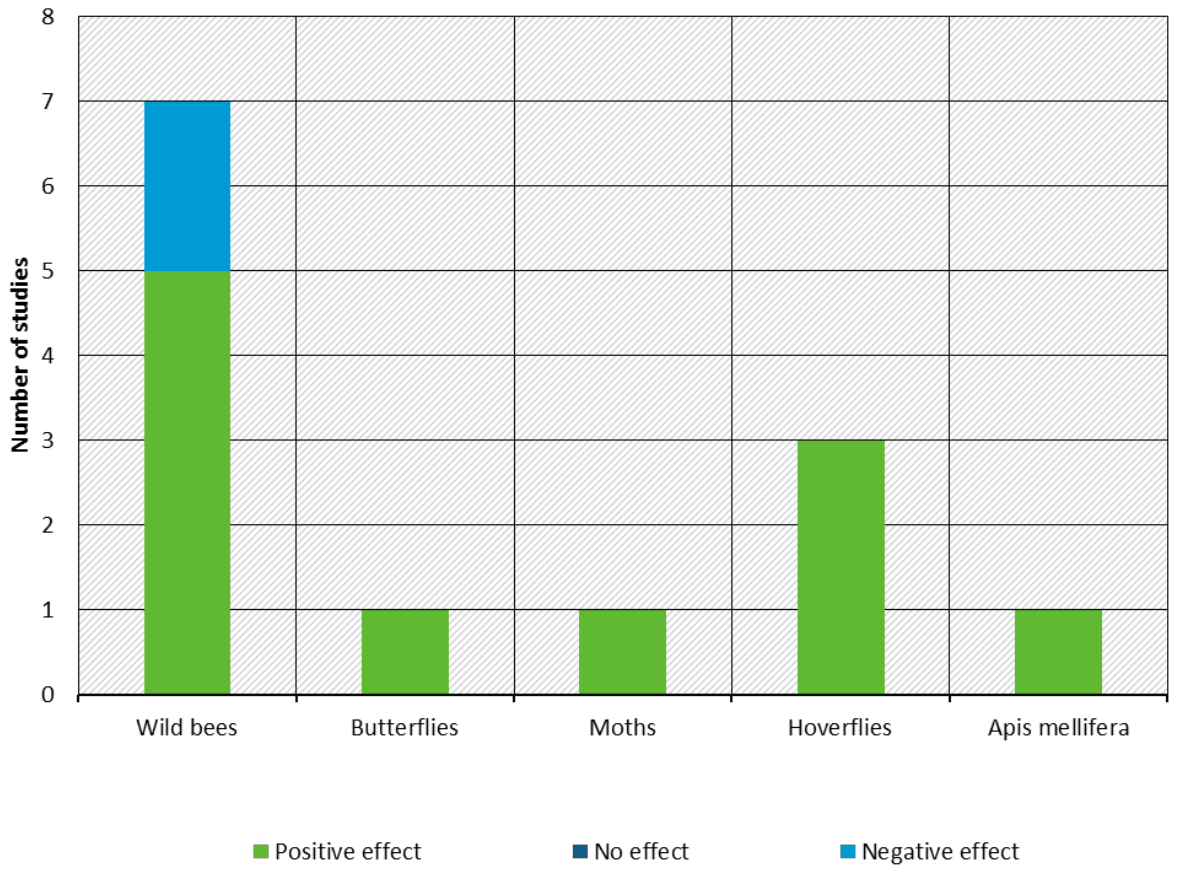
Footnote: Soderman et al. (2016) showed positive effects on hoverflies, no effects on wild bees and even negative effects on bumble bees. Therefore, this study is assigned to a fourth category (positive, no - and negative effect) of the bar chart.

Source: own illustration, ecotox consult

In general, the most investigated taxa are wild bees including bumble bees. The analysed studies showed positive significant effects of hedges on species richness or abundance of bumble bee (Croxton et al. 2002), wild bee (Hannon & Sisk 2009; Morandin & Kremen 2013; Kremen & M'Gonigle 2015), moth (Merckx et al. 2009), hoverfly (Morandin & Kremen 2013; Kremen & M'Gonigle 2015; Soderman et al. 2016) and *Apis mellifera* (Croxton et al. 2002). Furthermore, rare bee species as well as less mobile species were attracted by established hedges dependent on the available pollen and nectar supply (Hannon & Sisk 2009; Morandin & Kremen 2013). In total, there are seven studies in available literature investigating wild bees which include bumble bees (Figure 28). Five of them showed positive effects on abundance and/or species richness and two of them negative effects (no one showed no effects) (Figure 28). One study showed positive effects on butterflies, moths and *Apis mellifera*, respectively (no effects or negative effects were not found). Positive effects on hoverflies were found in three studies (no effects or negative effects were not found).

Most research focused on the effects of hedges on FVI populations. Only three studies investigated trees (Diekotter et al. 2006; Haaland & Gyllin 2010; Soderman et al. 2016). One of them showed positive effects on butterflies and bumble bees (Haaland & Gyllin 2010) and another showed positive effects on hoverflies (Soderman et al. 2016). In case of single trees without understorey, we assume that effects on FVI populations might possibly be lower than reported for hedges (e.g. due to less nesting sites). The results of the studies presented in (Table 46) prove the positive effects of high vegetation on different groups of FVIs. However, it has to be noted that the establishment of high vegetation (trees and hedges) in agricultural landscapes might also negatively impact certain FVI groups (e.g. bumble bees) which prefer rather open landscapes (Diekotter et al. 2006; Soderman et al. 2016) (see also Figure 28).

Figure 28: Number of available studies demonstrating effects (positive, no or negative) of hedges and trees on abundance or species richness of bees, butterflies, moths or hoverflies. Data is derived from Table 46.



Footnote: In the study by Soderman et al. 2016 only bumble bees were negatively affected by hedges. Other wild bees were unaffected (neither positive nor negative). Thus, the effect on the whole group of wild bees (including bumble bees and other wild bees) was evaluated as negative in the bar chart.

Source: own illustration, ecotox consult

In general, it seems that higher diversities of plant species (preferably perennial plants) in hedges provide nesting habitats of high quality (dead wood, woody vegetation) for different FVI species (Croxton et al. 2002; Hannon & Sisk 2009; Morandin & Kremen 2013). Furthermore, in hedges specific host plants can occur (e.g. *Dactylis* glomerata, *Lotus corniculatus* or *Urtica dioica*) which might increase species richness and abundance of butterflies (Dover & Sparks 2000).

The potential to attract FVIs seems to increase with increased maturation of the vegetated landscape element. It was shown by Kremen & M'Gonigle (2015) that the species richness of wild bees and hoverflies significantly increase with successional year due to the ripening of plant resources in hedges. Furthermore the occurrence of small FVI species (mainly cavity nesters) which are specialized in floral and nesting resources also increased with the maturation. Moreover, the vertical distribution of nesting locations changed over time which might result in a time-dependent change of FVI community composition; i.e. Kremen & M'Gonigle (2015) highlighted that with the age of the hedge abundance of above-ground nesting bees increased.

In addition, boundary strips between fields and the hedge can have additional positive effects on FVI species richness (Marshall & Moonen 2002) because FVI communities might be better protected against pollutants and pesticides. Kjaer et al. (2014) underlined that by increasing the distance between crops and hedge, pesticide spray drift entries in the hedge are reduced which might promote FVI populations. Hanley & Wilkins (2015) found that bumble bees were more abundant at the roadside margin of the investigated hedges than at the crop-side margins, which might be a result of less pesticide exposure at the roadside margin. Less herbicide exposure might promote the occurrence of floral nectar and pollen resources (which were found to be decreased at the crop-side), and less insecticide exposure might reduce potential direct effects to FVI.

The frequency of cutting hedgerows can have profound effects on the density of flowers, as many hedgerow species flower on second-year old growth (Holland et al. 2016). Thus, an influence on FVI communities and their diversity might be expected. For example, Maudsley et al. (2000) showed a tendency that Lepidoptera larvae were more abundant in May and July when the hedge is unmanaged or cut once in September compared to the cutting in February. However, the observed difference is only statistically significant at two of six sampling sites. The author assumed, the lower abundance after February cutting might be caused by removal of Lepidoptera eggs which are laid in the hedge during autumn through the winter.

8.2.4.3 Feasibility and acceptability by farmers

The planting of high vegetation as risk mitigation measure is not easy feasible, because the planting requires high effort (Bereswill et al. 2014). Furthermore, Reichenberger et al. (2007) pointed out, that the development of a full leaf stage, which is required for an effective mitigation of pesticide spray drift entries, takes many years. Farmers might be concerned about the creation of high vegetation adjacent to their agricultural land because shading of agricultural fields (Bereswill et al. 2014) or increase of pest/disease pressure (Reichenberger et al. 2007) might possibly occur. Furthermore, Bereswill et al. (2014) suggest that farmers might be worried about the creation of new inherently protected habitats when planting high vegetation. Therefore the acceptability by farmers to plant high vegetation as risk mitigation measure is assumed to be rather low.

In short: hedges, single trees, trees in a line and trees in a group

- ► High vegetation (such as hedge and shrubberies) in full leaf stage can provide a reduction of pesticide entries in habitats behind the hedge/shrubbery by 75% and more. Reduction efficiencies of single trees or trees in a row might be lower, in case that a close understorey lacks.
- ► Numerous studies prove that high vegetation in the agricultural landscape promotes FVI abundance and species richness (wild bees, bumble bees, hoverflies, and Lepidoptera). However, there are also negative impacts particularly on species which are mainly attracted by open landscapes.
- ► The attractiveness of hedges is mainly dependent on plant diversity (including the presence of host plant species) and the amount of available nesting opportunities.
- ► The implementation of this measure is difficult feasible because the planting requires high effort. Acceptability by farmers is presumably rather low, because yield losses (due to shading, increased pest pressure) might possibly occur, and farmers concern about the creation of new inherently protected habitats.

Table 46: Statistically significant effects of hedges and trees on flower-visiting insects (FVI) as investigated in field studies (n.r. = not reported).

Reference	Hedge/tree type	Hedge/tree size	Positive effect on FVI		Investigated taxa	Adjacent crop	Monitoring period
			Abundance	Species richness			
(Croxton et al. 2002)	Hedges	100 m long and height ranged from small (1 m) to tall and gappy (size is not available)	**Yes**	n.r.	Bumble bees *Apis mellifera*	Wheat Oil seed rape	1 year
(Diekotter et al. 2006)	Tree groups and trees in a line	n.r.	No[a]	Not relevant	*B. muscorum*	Agricultural fields	2 years
(Haaland & Gyllin 2010)	Greenways planted with trees and bushes	4 m wide	**Yes**[b]	n.r.	Bumble bees	Arable land	Planting in the 1990s; Recordings 2007
(Hannon & Sisk 2009)	Greenways planted with trees and bushes	4 m wide	**Yes**[b]	**Yes**	Butterflies	Arable land	Planting in the 1990s; Recordings 2007
(Hannon & Sisk 2009)	Hedges (riparian - velvet mesquite trees)	Not relevant	**Yes**[c]	**Yes**	Wild bees	Hay crops	2 years
(Kremen & M'Gonigle 2015)	Hedges (native perennial shrubs and trees)	350 m long and 3–6 m wide	**Yes**	**Yes**	Wild bees	Conventional row crops Vineyards Orchards	Recordings 6 year after hedgerow restoration
(Kremen & M'Gonigle 2015)	Hedges (native perennial shrubs and trees)	350 m long and 3–6 m wide	No	**Yes**	Hoverflies	Conventional row crops Vineyards Orchards	Recordings 6 year after hedgerow restoration

Reference	Hedge/tree type	Hedge/tree size	Positive effect on FVI		Investigated taxa	Adjacent crop	Monitoring period
			Abundance	Species richness			
(Merckx et al. 2009)	Open-grown hedge trees (predominantly *Quercus robur*)	Minimum height of 15 m	**Yes**	**Yes**	Moths	Agricultural fields	2 years
(Morandin & Kremen 2013)	Hedges	305 m to 550 m long	**Yes**	**Yes**	Wild bees	Agricultural fields	2 years
(Morandin & Kremen 2013)	Hedges	305 m to 550 m long	**Yes**	**Yes**	Hoverflies	Agricultural fields	2 years
(Soderman et al. 2016)	Hedges and trees	Not specified	No[d]	No[d]	Bumble bees	Cereals and Sugar beet	1 year
(Soderman et al. 2016)	Hedges and trees	Not specified	No	No	Wild Bees	Cereals and Sugar beet	1 year
(Soderman et al. 2016)	Hedges and trees	Not specified	**Yes**	No	Hoverflies	Cereals and Sugar beet	1 year

[a] The abundance was negatively influenced since *B. muscorum* is an open-landscape species.

[b] FVI abundance was significantly increased by grassy strips with trees and bushes, however, the abundance was approximately 20 times smaller than in wildflower strips.

[c] Species richness and abundance was highest at pre-monsoon season (early summer).

[d] The removing of hedges and trees positively significantly influenced the abundance and species richness of bumble bees.

8.2.5 Reduction of application rate and application frequency and modifications of intervals between applications

8.2.5.1 Reduction of pesticide entries and reduction of exposure of FVI in-crop

The reduction of the pesticide application rate is commonly proposed to reduce pesticide inputs via spray drift in off-field habitats (SANCO 2002; FOCUS 2007; Reichenberger et al. 2007; Bereswill et al. 2014). Moreover, because less pesticides are applied on the agricultural land, also the pesticide exposure of FVI in-crop (e.g. via overspray, foraging pollen) is reduced. The pesticide reduction efficiency corresponds to the extent of the reduced application rate (Reichenberger et al. 2007; Bereswill et al. 2014) and is in general linearly proportional to application rate (FOCUS 2007).

The reduction of application frequency results in a lower number of pesticide application events. Consequently, the amount of pesticides applied on agricultural fields is reduced, which also results in lower pesticide entries in off-field habitats. Therefore, via reducing the application frequency the exposure of organisms off-crop as well as in-crop might be also reduced (Alix et al. 2015), especially in the context of long-term exposure.

8.2.5.2 Effects on FVI

The reduction of pesticide application rate and application frequency result in a reduced pesticide exposure of FVI in off-crop and in-crop areas, and thus, potential effects of pesticides on FVI populations might decrease. The extent of this decrease is highly dependent on the specific design of these measures (e.g. amount of the reduced application rate) and on the specific pesticide. Also the extension of the interval between two pesticide applications can be a crucial factor influencing the effects on FVI. However, the actual influence is highly substance specific (e.g. persistence, toxicity of the pesticide) and highly specific for the FVI specie of concern (species traits), therefore the effect of this measure cannot predicted for FVIs in general.

8.2.5.3 Feasibility and acceptability by farmers

The application rate, frequency and interval can only be modified in the way that the efficiency of the specific pesticide use is still ensured. The feasibility is highly dependent on the pesticide and the respective pest pressure (Bereswill et al. 2014). If an efficient use with modified application rate, frequency or intervals is demonstrated, the measure might be accepted by farmers.

8.2.6 No-spray zones

No-spray zones or no-spray buffers are defined as an (in general cropped) area of certain width at the edge of a field where a specific pesticide is not allowed to be applied. The measure is currently already in force in Germany (as also in many other EU-member states (FOCUS 2007)) and is applied in the pesticide risk authorization process for a specific pesticide use. The efficiency of no-spray zones to reduce pesticide spray drift entries in adjacent habitats is well investigated (Ganzelmeier et al. 1995; Rautmann et al. 2001; Van de Zande et al. 2015) and corresponds to the values described for in-field buffer strips (section 8.2). In addition to spray drift, also the pesticide exposure of FVI in-crop, i.e. in the area of the no-spray zone, is reduced. However, there will be no effect in the other parts of the field.

8.2.6.1 Effects on FVI

With respect to no spray zones and their effects on FVI, only studies could be found in literature which investigated conservation headlands (unsprayed field edges), i.e. cropped buffers where the application of pesticides is generally not allowed. These studies (de Snoo et al. 1998; de Snoo 1999; Pywell et al. 2005; Carvell et al. 2007) were already included in the literature analysis of in-field buffer strips (section 8.2.1.2; Table 43).

Most of these studies focus exclusively on the effects of conservation headlands on FVI in-field, i.e. the abundance of FVI on the conservation headlands were compared to a cropped field control (de Snoo et al. 1998; Pywell et al. 2005; Carvell et al. 2007). Carvell et al. (2007) and Pywell et al. (2005) did not found any significant differences between bumble bee abundance and richness on conservation headlands and controls (conventional cereal field edge and crop, respectively). In contrast, de Snoo et al. (1998) found significant higher butterfly abundance and species richness in 3-6 m wide unsprayed winter wheat field edges in comparison to sprayed field edges. Furthermore, a three to four times higher insect density (mainly FVI such as hoverflies and aphid predators such as ladybirds) was found in the upper parts of the vegetation of the unsprayed field edges in comparison with sprayed field edges (de Snoo 1999).

de Snoo et al. (1998) also surveyed butterflies at ditch banks bordering unsprayed field edges. They observed a positive effect on butterfly populations in the adjacent off-field habitat: in most cases number of taxa and abundance of butterfly species significantly increased also at the ditch bank bordering unsprayed field margins.

In conclusion the data base concerning field studies investigating the effects of no spray zones on FVIs is rare and further research regarding this topic is needed. The few studies available indicate that there might be positive effects on abundance and/or species richness of single groups of FVI such as butterflies in no-spray zones. However, this observation could not be confirmed by field studies for other groups such as bumble bees. Nevertheless, because pesticide spray drift is reduced by the implementation of no spray zones and thus, the exposure of FVI off-field is reduced, it can be assumed that there might be positive effects on FVI populations in the off-field area. This assumption was for example confirmed by de Snoo et al. (1998) for butterflies.

8.2.6.2 Feasibility and acceptability by farmers

No-spray zones can in general easily be implemented by farmers (Bereswill et al. 2014) by closing the drift nozzles at the field edge. However, because losses in crop yields might occur acceptability might partly be low as also assumed by Schulz et al. (2009). This might lead consequently also to problems in compliance with these regulations.

In short: no-spray zones

- ► Reduction of pesticide entry in off-field habitats by no-spray zones is dependent on the width of no-spray zones and decreases exponentially with increasing width.
- ► The data base concerning field studies investigating the effects of no-spray zones on FVIs is rare. However, there are indications that there might be positive effects on abundance and/or species richness of certain groups of FVI in in-field areas (e.g. butterflies).
- ► Positive effects on FVI in the off-field area can be assumed because pesticide spray drift is reduced by the implementation of no-spray zones and thus, the exposure of FVI in off-field habitats is reduced.
- ► Implementation of no spray zones seems to be feasible. Acceptability by farmers might be low.

8.2.7 Spray drift reducing techniques

8.2.7.1 Reduction of pesticide entries

With regard to spray drift reducing techniques there are different possibilities to reduce pesticide spray drift during the application process such as conventional sprayers equipped with drift reducing nozzles and/or air assistance or specific recycling sprayers e.g. tunnel sprayers. By using spray drift reducing nozzles, spray drift can be reduced up to 75-90% and more (Koch et al. 2003; Van de Zande et al. 2008; Schweizer et al. 2013). The achieved reduction efficiency depends on the used nozzle type, the spray pressure and the resulting spray quality. In general, the coarser the resulting spray droplets the lower is the spray drift.

Van de Zande et al. (2008) further showed that the use of air assisted field sprayers provides a reduction of pesticide spray drift around 70% independent of the used nozzle type. However, Schmid (2001) underlines that the use of air assistance in field spraying might possibly increase pesticide spray drift when no vegetation or only small plants were present on the field.

Recycling sprayers such as tunnel sprayers are specifically developed for the application of pesticides in vineyards, fruit crops or hops. For these sprayers, spray drift reduction values around 80-90% are reported (Schmid 2001; Van de Zande et al. 2008; Doruchowski et al. 2013). For the spraying in vineyards and fruit crops there are also sensor controlled sprayers available. Nozzles can be opened and closed controlled by the signal of a sensor which can identify gaps in the sprayed canopies. Thus, reduction efficiencies in the range of 25-50% might be achieved (Doruchowski & Holownicki 2000; Schmid 2001).

Spray drift reducing techniques are currently implemented in the pesticide risk authorization process in Germany and many other EU-Member-States (e.g. Austria, Netherlands, UK). In Germany, drift reducing techniques are typified according their drift reduction in classes of 50%, 75% and 90% and are listed on the official list of spray drift reducing techniques (JKI 2013).

A further reduction of pesticide exposure of FVIs in-crop does not occur as shown in Table 42.

8.2.7.2 Effects on FVI

The reduction of pesticide spray drift in non-target terrestrial habitats results in a lower pesticide exposure of FVI within these areas. In consequence, the risk for potential pesticide effects on FVI populations should decrease.

8.2.7.3 Feasibility and acceptability by farmers

Provided that appropriate spraying techniques (e.g. drift reducing nozzles) are available at farms, the implementation of this mitigation measure seems to be feasible and the acceptability by farmers is assumed to be high. Especially because the measure rather achieves that the amount of applied pesticides reaches the target location and losses of crop yield are not expected.

In short: spray drift reducing technique

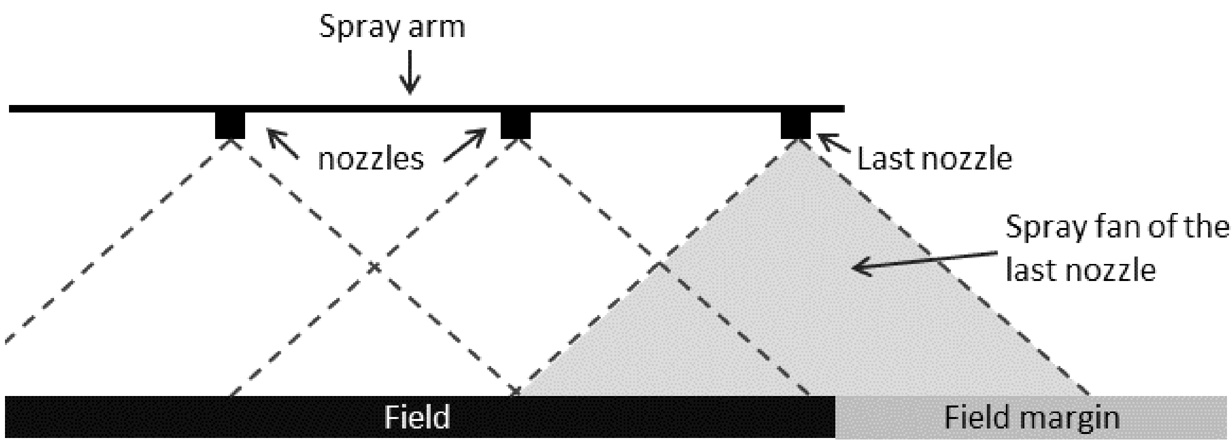
- ► Different spray drift reducing techniques are available: e.g. spray drift reducing nozzles, tunnel sprayer, sensor controlled sprayer.
- ► Depending on the chosen technique pesticide spray drift reductions of 25-90% or more can be achieved.
- ► Spray drift reducing techniques are currently implemented in the pesticide risk authorization process in Germany and many other EU-Member-States.
- ► Provided that appropriate spraying techniques (e.g. drift reducing nozzles) are available at farms, the implementation of this mitigation measure seems to be feasible and the acceptability by farmers is assumed to be high.

8.2.8 No overspraying of off field habitats

8.2.8.1 Reduction of pesticide entries

According the German plant protection act (paragraph 12, section 2) plant protection products are only allowed to be applied on agricultural, forestry or horticultural areas. On other areas such as terrestrial off field habitats plant protection products must not be applied. However, an overspraying of these areas can indeed occur. As described by Schmitz et al. (2013) when spraying field crops, the nozzles are positioned on the sprayer boom so that the spray cones of two nozzles overlap which results in a 100% application rate. In consequence the last nozzle which is located above or near the field edge also oversprays partly the adjacent field margin. For example using fan nozzle with a spraying angle of 110° as the last nozzle on the spray arm and a spraying height of 50 cm, ca. 70 cm of the adjacent field margin is oversprayed (Koch 2009) with 50% of the application rate.

Figure 29: Schema illustrating the occurrence of pesticide entries via overspray in a field margin during a pesticide application in field crops (adapted from (Schmitz et al. 2013)).



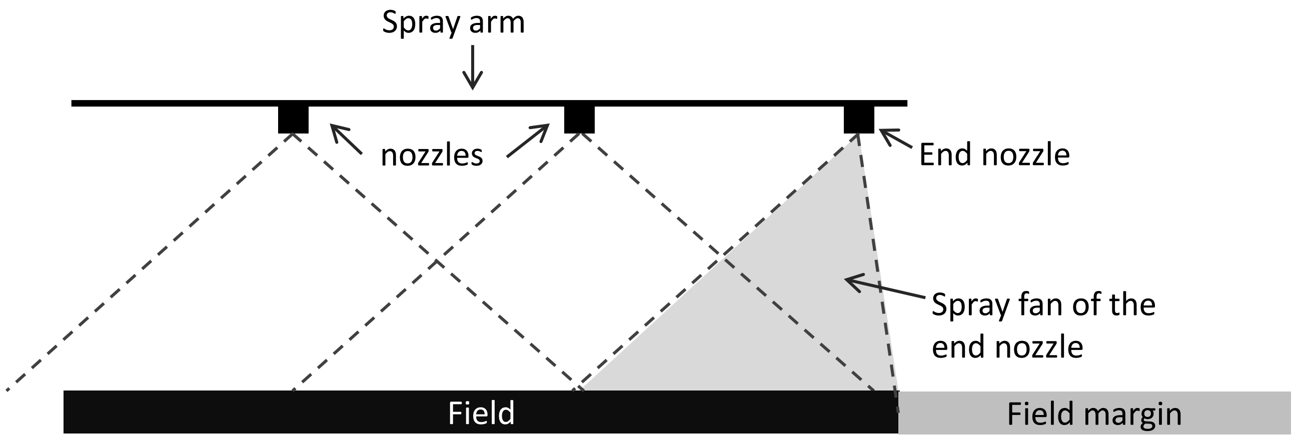
Source: own illustration, ecotox consult (adapted from Schmitz et al. 2013)

To avoid this kind of overspray, several options are available. First, the implementation of a mandatory standard distance of e.g. 1 m between the last spraying nozzle and the adjacent field margin might be suitable. This could be easily achieved by closing the last spraying nozzle(s) or by increasing the distance of the pesticide application device and the field margin. Also in case that in-field buffer strips (such as described in section 8.2) are implemented, the occurrence of oversprayed off-field habitats can be avoided.

Another possibility to reduce overspray of filed margins is the use of specific end nozzles (BMEL 2017). The spray fan of an end nozzle is cut-off in the direction of the field margin (Van de Zande et al. 2008). The functional principle of an end nozzle is exemplarily shown in Figure 30. The German BVL underlines in an official notice in the German Bundesanzeiger in 2013 that the overspray of off-field habitat is prohibited and that the use of end nozzles is appropriate to avoid this type of overspray (BVL 2013). Koch (2009) further underlines that by using end nozzles also the spray drift in the adjacent immediate vicinity of the field margin (first 1-3 m) can be additionally reduced. The combination of an end nozzle (UB03) with the nozzle XR 110 03 (not drift reducing) provided a drift reduction at 1 and 3 m distance of 74% and 62%, respectively. According to Van de Zande et al. (2008) the end nozzle (UB8504) in combination with a low-drift nozzle (DG11004) resulted in a spray drift reduction of 50% at 1-2 m distance and of 20% at 2-3 m distance. The reduction of pesticide exposure in the first 1-3 m of the field margin is particularly important because most field margins are narrow (< 3 m) in intensively used agricultural regions (Hahn et al. 2014).

A reduction of pesticide exposure of FVIs in-crop does not occur by avoiding the overspray of off-field habitats (Table 42).

Figure 30: Schema illustrating the effect of end nozzles during a pesticide application in field crops.



Source: own illustration, ecotox consult

8.2.8.2 Effects on FVI

The reduction of pesticide overspraying of off-field habitats should promote FVI populations in these areas. On the one hand the pesticide exposure of FVI in the direct vicinity to the field is reduced. On the other hand also the exposure of field margin plants to herbicides is reduced which should support the occurrence of food plants for FVIs and thus, indirectly positively affect FVI populations.

8.2.8.3 Feasibility and acceptability by farmers

The implementation of this measure seems to be feasible. Conventional sprayers can be equipped with appropriate end nozzles, or the last nozzles can be closed to increase the distance between last spraying nozzle and the adjacent field margin. Provided that appropriate spraying techniques are available at farms, acceptability by farmers at least for the use of end nozzles is assumed to be high, because losses of crop yields are not expected in consequence of this method. Acceptability of the implementation of a standard distance between the sprayed area and the field might be lower, because in this case loss of crop yields at the field edge might occur.

In short: no overspraying of off-field habitats

- Pesticide overspray can be mitigated by the use of specific end nozzles. In addition, by the use of end nozzles the spray drift in the adjacent immediate vicinity of the field margin (first 1-3 m) can be efficiently reduced. Reduction values of 20-74% are reported.
- Another possibility to reduce overspray of off-field habitats is the implementation of a compulsory standard distance (e.g. 1 m) between the last spraying nozzle and the adjacent field margin.
- Implementation of this measure seems to be feasible. Provided that appropriate spraying techniques are available at farms, acceptability by farmers for the use of end nozzles is assumed to be high, but might be lower for the introduction of a mandatory standard distance.

8.2.9 Timing of application

A further possibility to reduce the pesticide exposure of FVI on in-crop areas is the appropriate timing of the application. Pesticides hazardous to bees or other diurnal FVIs should only be applied in the evening after flight of these species. This risk mitigation measure is already in force in Germany: Pesticides classified as hazardous to bees with the label B2 are not allowed to be used on plants which are in flowers or which are visited by bees except after 11 p.m. when daily bee flight is finished (BVL 2016). However, applying this risk mitigation measure might increase the risk for FVI groups which are active in the night (e.g. moth). Moreover, the exposure of FVI towards pesticides can further be mitigated, when pesticides are not applied when crops flowers or flowering vegetation between fruit tree/vineyard rows are present. In these cases, pesticides less toxic to FVI should be chosen for application or alternative pest control techniques should be chosen. This measure is currently also in force in Germany for protecting bee populations: Pesticides classified as hazardous for bees with the label B1 are not allowed to be used on plants which are in flowers or which are visited by bees (BVL 2016). Depending on the specific pesticide active ingredient, the restriction of the application when crops flowers or flowering vegetation is present between fruit tree/vineyard rows might be highly effective to reduce the direct exposure of diurnal FVIs via overspray and spray drift.

Pesticide spray drift entries in off-field habitats can generally be mitigated when pesticides are applied only at low wind speeds. According to the rules of good agricultural practice wind speed should be less than 5 m/s during application. The lower the wind speed , the lower the amount of pesticide spray drift in off-field habitats (Van de Zande et al. 2015).

8.2.10 Preservation and management of existing off-field habitats

Already existing off-field habitats (e.g. meadows, field margins, hedges, shrubs) in the agricultural landscape should be preserved and managed with aim a) to provide diverse habitats for FVI populations, b) to provide appropriate food resources during the whole season and c) to ensure good opportunities for nesting and reproduction. Consequently, these areas might stabilize the presence of FVI populations in the agricultural landscape and act as refuge habitat or recolonization source for FVI populations.

Attractiveness of existing off-field habitats (e.g. field margins, meadows) can be increased for FVI species by sowing specific seed mixtures. As outlined in section 8.2 and 8.2.3 by increasing the number and diversity of forage flowers in off-field habitats, the presence of FVI species is promoted (Carvell et al. 2007; Kohler et al. 2008; Kuussaari et al. 2011; Gill et al. 2014; Scheper et al. 2015). Moreover, the presence of larval host plant species is of great importance for FVI groups such as Lepidoptera (Holland et al. 2015).

Time and rhythm of mowing of off-field habitats such as field margins or meadows should be optimized with regard to protection of FVIs. Plattform Bienenzukunft (2016) recommends postponing the time of mowing in case of a high bee activity on the vegetation (> 1 bee/m^2). However, the scientific database of this recommendation is not clear. When mowing, some parts of the vegetation (minimum 10-30% of the area is recommended) should be left in flowers as isles of flowering vegetation (Oppermann et al. 2013; Fenchel et al. 2015; Plattform Bienenzukunft 2016). Fenchel et al. (2015) further recommends for the preservation of a perennial flower strip that such a mowing should take place mid/end of July in at least 15 cm height. The mowed vegetation might have a second bloom which provides important food resources for bees later in summer/autumn (Oppermann et al. 2013; Fenchel et al. 2015). However, disturbances in consequence of mowing should be as far as possible reduced and intervals between mowing should be as long as possible (Nitsch et al. 2016). Also Halbritter et al. (2015) concluded based on their studies of the effect of mowing frequency (no mowing, every 6 weeks, every 3 weeks) of roadside margins on butterfly abundance, that reducing or avoiding mowing during peaks of butterfly activity might be beneficial. The mowing in three week intervals yielded the lowest flower abundance and species richness. However, a significant effect of mowing alone on butterfly abundance was not observed. But inspection of population dynamics by expert judgement indicated a tendency that the no-mow treatment showed the highest butterfly abundance in August and onwards. Similarly, also Saarinen et al. (2005) and Bak et al. (1998 in: Halbritter et al. 2015) found that high mowing frequency negatively affected butterfly abundance in Finland and New Zeeland, respectively. Nevertheless, mowing might partly also be necessary in the context of flower strip maintenance to manage undesirable plant species upcoming in flower strips as described for example by Stückrath et al. (2013) and Beyer et al. (2013). Please refer also to the discussion of mowing activities in fallows (section8.2.3.2).

In addition, also the maintenance of hedges present on off-field habitats can be managed with regard to the protection of FVIs. Plattform Bienenzukunft (2016) recommends that dominant plant species such as hazel should be cut back, whereas important food resource taxa for FVIs (e.g. hawthorn, blackthorn, shrub willows) should be preserved. This approach allows maintaining a hedge composed of diverse plant species providing food and nesting habitats for FVIs. However, cutting of hedges should be reduced in general, because cutting result in a reduced number of flowers. Staley et al. (2012) found that reducing the cutting frequency from annually to every 3 years resulted in 2.1 times more flowers. Hence, it could be assumed that food resources for FVIs are increased.

To support nesting and reproduction of FVIs, appropriate additional nesting habitats off-field should be created. According to Plattform Bienenzukunft (2016) wood piles or piles consisting of stones (see also ecological focus area dry stone walls) and sand might be suitable structures. The cavities of such structures might provide important nesting habitats for certain cavity nesting bee species so called renter (please refer to section 0). However, detailed field studies investigating the extent of the effect of created nesting structures such as piles consisting of stones and wood on FVI populations are currently not available.

8.2.11 Leaving deadwood in fruit orchards

Plattform Bienenzukunft (2016) also recommends to left deadwood (i.e. dead branches on old trees, or dead trees) in fruit orchards as long as possible. These structures provide important nesting habitats for certain cavity nesting bee species so called renter (please refer to section 0), and might thus improve stability of bee populations in agricultural landscapes. For this, retaining deadwood is also recommended in the context of entry and higher level stewardship schemes (agri-environment programmes) in England (Anonymous 2013a; Anonymous 2013b). However, detailed field studies investigating the extent of the effect of deadwood on FVI populations are currently not available. With this respect there is need for further research.

Leaving dead branches or dead trees in fruit orchards requires not much effort and is therefore assumed to be easy feasible. Acceptability is assumed to be moderate to high, at least in case that workers safety and passage of tree rows with machines is provided.

8.2.12 Sowing of seed mixes between vine and fruit tree rows

In addition also the sowing of flower seed mixes between vine and fruit tree rows might promote FVI populations in agricultural landscape. An increase of forage flower number and diversity promotes the presence of FVI species (Carvell et al. 2007; Kohler et al. 2008; Kuussaari et al. 2011; Gill et al. 2014; Scheper et al. 2015) by providing additional food resources. However, in case of flowering vegetation between vine and fruit tree rows, farmers should pay special attention when applying pesticides which might be toxic for FVI. As already outlined in section 8.2.9 pesticide application should then take place in the evening when diurnal FVI are not active. Another possibility might be the mowing of the flowering vegetation before pesticide application.

This measure is easy feasible, at least the sowing of seed mixes in every second vine or fruit tree row. Acceptability might be moderate to high, because no yield losses or loss of land for agricultural production is expected, but there might be additionally maintenance effort (e.g. mowing).

8.2.13 Overview of the efficiency of evaluated risk mitigation measures to promote flower-visiting insects

The efficiency of the evaluated risk mitigation measures to promote FVIs as discussed in the previous sections is summarized in Table 47. The table shows the number of studies (treatments, respectively) available in literature reporting positive effects on FVIs, no effects or negative effects. Moreover, the conclusion drawn based on the available data and information discussed in the previous sections is presented.

Best investigated mitigation measures are wildflower in-field buffer strips and fallows. For both the efficiency to promote FVIs was scientifically demonstrated in available literature. In addition efficiency could also be demonstrated for further risk mitigation measures (e.g. hedges, sowing of seed mixtures).

However, there are many measures with further need of research. Particularly in case of application related measures (e.g. reduction of application rate, spray drift reducing techniques) there are no studies available investigating the effects on FVIs. In general, focus of available studies is the potential to reduce pesticide entries in off-field habitats or pesticide inputs in-crop. Based on the efficiency of these measures to reduce pesticide exposure, it can be assumed that also effects on FVIs are reduced. But there are also some landscape-related measures (e.g. creation of nesting possibilities, leaving deadwood in fruit orchards) where the available database in literature is insufficient. However, efficiency to promote FVIs can be assumed based on ecological considerations.

In case of landscape related risk mitigation measures (e.g. buffer strips, hedges) the question raises if these measures should be additionally protected from pesticide entries. This question is comprehensively discussed for ecological focus areas in section 8.4.5. For ecological focus areas which have the main aim to protect biodiversity, we concluded that EFAs should be protected from pesticide entries such as pesticide drift (section 8.4.5) (e.g. by no spray zones or uncultivated buffers around EFAs, or using end nozzles). If landscape related risk mitigation measures represent natural biotopes/landscape elements already present in the agricultural landscape (e.g. natural hedges, field margins), the same conclusion has to be drawn for landscape risk mitigation measures. In case that landscape related risk mitigation measures are specifically created/implemented to reduce effects on FVIs by acting as a buffer for pesticides (e.g. in-field buffer strips) and/or promote FVIs in-field (e.g. sowing of seed mixes between vine and fruit tree rows) it might be justifiable to discuss if same requirements hold for these measures than for natural biotopes. It is obvious that greatest positive effects on FVI biodiversity will occur when all landscape related measures (natural biotopes as well as measures specifically implemented for risk reduction) are protected from pesticide entries. However, a landscape with e.g. a high amount of in-field buffer strips might be at least more efficient in promoting FVIs than a landscape without such elements, even when in-field buffers might not be further protected against spray drift entries.

Table 47: Overview of proposed risk mitigation measures and their efficiency to promote flower-visiting insects (FVIs) (- no studies available)

Risk mitigation measure		Number of studies (or treatments, respectively)[a] in available literature reporting			Efficiency to promote FVIs
		positive effects on FVI	no effects on FVI	negative effects on FVI	
In-field buffer	Wildflower	11[(1)]	0	0	Demonstrated
	Conservation headland	3[(2)]	2[(3)]	0	Efficiency demonstrated, but in case of wild bees the efficiency is assumed to be limited [b]
	Grassy	4[(4)]	2[(5)]	0	Efficiency demonstrated, but in case of wild bees the efficiency is assumed to be limited [b]
	Natural regeneration	1[(6)]	2[(7)]	0	Inconsistent database, but efficiency can be assumed based on ecological considerations
Extension of small field margins		3[(8)]	0	0	Demonstrated
Conservation fallow		14[(9)]	1[(10)]	0	Demonstrated
High vegetation (hedges/trees)		7[c,(11)]	1[c,(12)]	2[c,(13)]	Demonstrated, but in case of single trees without understorey lower efficiency than for hedges is assumed.
Reduction of application rate/frequency		-	-	-	Insufficient database, but efficiency can be assumed due to reduction of pesticide exposure
No-spray zones		3[d,(2)]	2[d,(3)]	0	Efficiency demonstrated, but in case of wild bees the efficiency is assumed to be limited
Spray drift reducing techniques		-	-	-	Insufficient database, but efficiency can be assumed due to reduction of pesticide exposure
No overspraying of off field habitats		-	-	-	Insufficient database, but efficiency can be assumed due to reduction of pesticide exposure
Timing of application		-	-	-	Insufficient database, but efficiency can be assumed due to reduction of pesticide exposure

Risk mitigation measure		Number of studies (or treatments, respectively)[a] in available literature reporting			Efficiency to promote FVIs
		positive effects on FVI	no effects on FVI	negative effects on FVI	
Preservation and management of existing off-field habitats	Sowing of seed mixtures	5[e,(14)]	0	0	Demonstrated.
	Mowing rhythm (prolonging interval between mowing)	2[(15)]	1[f,(16)]	0	Efficiency demonstrated with limitations for butterflies. However for other FVI taxa efficiency can be assumed based on ecological considerations.
	Maintenance of hedges	-	-	-	Insufficient database, but efficiency can be assumed based on ecological considerations.
	Creation of nesting possibilities	-	-	-	Insufficient database, but efficiency can be assumed based on ecological considerations.
Leaving deadwood in fruit orchards		-	-	-	Insufficient database, but efficiency can be assumed based on ecological considerations.
Sowing of seed mixes between vine and fruit tree rows		5[e,(14)]	0	0	Demonstrated.

[a] In case of wildflower strips and fallows the different treatments in the available studies were counted, because there were authors who investigated different treatment of wildflower strips and fallows, respectively (see also Figure 24 and 26).

[b] In case of wild bees the efficiency is assumed to be limited (see Table 17).

[c] Sodermann et al. (2016) showed positive effects on hoverfly abundance, no effects on wild bee abundance and species richness and even negative effects on abundance and species richness on bumble bees. The study of Sodermann et al. (2016) is therefore counted in each column.

[d] For no spray zones only studies could be found in literature which investigated conservation headlands (unsprayed field edges), i.e. cropped buffers where the application of pesticides is generally not allowed. In contrast, buffer zones are typically stipulated for specific pesticide uses.

[e] Mentioned studies showed that by increasing the number and/or diversity of forage flowers in habitats, the presence of FVI species was promoted.

[f] Study of Halbritter et al. (2015) found no statistical significant effects of reducing mowing frequency on butterfly abundance, but a tendency was observed that the no-mow treatment showed the highest butterfly abundance in August and onwards.

(1) Aviron et al. 2007, Carvell et al. 2007 (investigated two treatments, i.e. wildflower strip and strip with nectar & pollen mixture), Kohler et al. 2008, Haenke et al. 2009 (investigated two treatments, i.e. 3-6 m wide flower strip and 12-25 m wide flower strip), Meek et al. 2002 (wildflower/grass strip), Oppermann et al. 2016; Pywell et al. 2005, Scheper et al. 2015, Wood et al. 2015.

(2) de Snoo et al. 1998 (investigated two treatments, i.e. conservation headland adjacent to winter wheat and conservation headland adjacent to potatoes), de Snoo 1999

(3) Carvell et al. 2007, Pywell et al. 2005

(4) Field et al. 2005, Field et al. 2007, Haenke et al. 2009, Marshall et al. 2006

(5) Carvell et al. 2007, Meek et al. 2002

(6) Pywell et al. 2005

(7) Carvell et al. 2007, Meek et al. 2002

(8) Backman & Tiainen 2002, Cole et al. 2015, Rands & Withney 2011

(9) Alanen et al. 2011 (investigated three different treatments), Denys & Tscharnke 2002, Diekotter et al. 2006, Gathmann et al. 1994 (two of three investigated treatments showed positive effects), Holland 2015, Kohler et al. 2008, Kuussaari et al. 2011 (investigated three treatments), Steffan-Dewenter & Tscharntke 2001 (investigated two different treatments)

(10) Gathmann et al. 1994 (one of three investigated treatments showed no effect)

(11) Croxton et al. (2002), Haaland & Gyllin 2010, Hannon & Sisk 2009, Kremen & M`Gonigle 2015, Merckx et al. 2009, Morandin & Kremen 2013, Sodermann et al. 2016 (positive effects were observed on hoverfly abundance)

different treatments)

(12) Sodermann et al. 2016 (no effects were observed on wild bee abundance and species richness)

(13) Diekotter et al. 2006, Sodermann et al. 2016 (negative effects were observed on bumble bee abundance and species richness)

(14) Carvell et al. 2007, Gill et al. 2014, Kohler et al. 2008, Kuussaari et al. 2011, Scheper et al. 2015

(15) Bak et al. 1998 (in: Halbritter et al. 2015), Saarinen et al. 2005

(16) Halbritter et al. 2015

8.3 Source-sink dynamics in agricultural landscapes and implications for risk mitigation measures

An important issue when considering the protection of FVI populations in agricultural landscapes is the spatial mobility of these organisms and resulting source-sink dynamics. For example, pesticide treated fields or off-field habitats receiving high pesticide entries might act as sink for FVI populations, whereas natural or semi-natural off-field habitats might ensure recovery of these areas and act as a source. The consequences of such dynamics are not always easy to predict (Topping et al. 2015).

Franzen & Nilsson (2013) investigated the population dynamics of a rare solitary bee species in Sweden (*Andrena humilis*) in a forest-dominated landscape of 80 km^2. *Andrena humilis* is an oligolectic bee specialized on a few pollen plant species; suitable habitats are grassland patches. Twelve habitat patches were observed with one large persistent population which seemed to act as a source which supplies the landscape with smaller temporary, non-persistent populations (in sink habitat patches). The authors observed population decreases in sink patches. They assume that in sink patches reproductive rates might be lower because less preferred pollen is harvested in these patches or distances between food resources and nesting sites are large. As expected, occupancy of sink patches was higher in patches close to the persistent population. This relationship is also confirmed by Jauker et al. (2009) who showed that wild bee abundance significantly decreased with increasing distance from the main semi-natural habitat (grassland). Furthermore, wild bee diversity showed the same significant relationship at least in landscapes with low-matrix quality (landscape with less than 10% grassland cover). Moreover, Franzen & Nilsson (2013) showed that the presence of high pollen resources[6] in sink patches buffered populations in these locations against extinction.

The results of Jauker et al. (2009) and Franzen & Nilsson (2013) underline the importance of large off-field habitats such as meadows, nature reserves or forests in the agricultural landscape which can act as a source for FVI populations. From these habitats recovery of smaller off-field habitats such as field margins or hedges (which are supposed to be more influenced by the agricultural practice such as pesticide applications) can take place. Therefore, large off-field habitats should be preserved in size and quality. This is also confirmed by Krewenka et al. (2011) who concluded based on their studies that large semi-natural habitats such as grasslands (in their case) should be protected as sources of bee and wasp diversity. Furthermore, to allow successful colonization it is important that the main source habitat is not too distant from the smaller off-field habitats, because Franzen & Nilsson (2013) showed that FVI populations were more stable in habitats close to the main habitat.

Furthermore, management of vegetation (e.g. sowing of seed mixes, modifications of mowing activities) in habitats which might act as a sink for FVI populations might be a useful tool. Franzen & Nilsson (2013) showed, high pollen resources buffered sink populations against extinction. Also Krewenka et al. (2011) assume that increasing the quality of e.g. grass strips by appropriate management measures might support bee and wasp diversity. In general sink habitats should be managed with respect to improve the quality of existing habitats and to create new habitats in order to provide resources for growth, maintenance and reproduction of FVI populations.

Examples for potential vegetation management measures:

- Mitigation measures such as in-field wild flower strips or sowing of seed mixes on present field margins.
- Creation of nesting structures in smaller off-field habitats such as field margins (e.g. by planting hedges or by creating nesting structures consisting of wood, stones or sand) as well as in in-field areas (e.g. by leaving deadwood in fruit orchards).

[6] High pollen resources were classified by the authors as more than 50 flowers observed in 10 squares (each of 0.5 m * 0.5 m).

Successful application of these vegetation management measures in sink habitats might positively influence population dynamics of FVIs. In consequence the impact of sinks habitats on FVI populations might be reduced. In best-case the managed sink habitats might even act as a source.

To generally reduce possible sinks of FVI in the agricultural landscape, it is important to support a FVI-friendly crop cultivation. These measures may include reduction of pesticide applications, the use of spray drift reducing techniques, a FVI-friendly timing of the application or organic farming. Particularly in case of mass-flowering-crops such as oilseed rape, which is known to attract FVI from (semi)natural habitats (Holzschuh et al. 2016), farmers should pay attention to use as far as possible pesticides non-hazardous to bees (and other FVI species) and to avoid spraying of hazardous pesticides during flight of FVI. Otherwise, it might be possible that FVI attracted by these crops are at risk for adverse effects as a consequence of pesticide applications.

To promote the long-term persistence of FVI populations in the agricultural landscape a preferably heterogeneous landscape consisting of a balanced mixture of different off-field habitats (e.g. grasslands, field margins, hedges, flower strips, fallows) and agricultural fields cultivated with different crops might be useful. For example, Jauker et al. (2009) found that bee species richness significantly decreased with increasing distance to the main grassland habitat in agricultural landscapes with a low matrix quality (i.e., less than 10% grassland cover in a buffer of 250 m around the sampling transect). However, no adverse impact were observed in landscapes with medium or high matrix quality (i.e., more than 10% grassland cover in a buffer of 250 m around the sampling transect). Rundlof et al. (2008) observed that bumble bee abundance and species richness was significantly higher in heterogeneous (mixed farmland) than in homogenous (intensive farmland) landscapes. Management measures implemented to achieve heterogeneity should include both linear structures (e.g. flower strips, field margins) and areal structures (e.g. fallows). Linear structures provide connections between off-field habitats, which is particularly important for organisms which are highly affected by habitat isolation as for example shown by Holzschuh et al. (2010) for wasps. Areal structures might be of particular importance because in large areal habitats boundary effects are reduced (e.g. boundary effects caused by pesticide drift entries).

However, certain FVI groups show a reversed pattern of the normally expected source-sink relationship, i.e. (semi-)natural habitat acting as a source and fields acting as a sink. Jauker et al. (2009) observed that hoverfly abundance increased in field margins with increasing distance to the main semi-natural habitat. This finding was explained to be a result of the special requirements of hoverfly larval stages. Hoverfly abundance was dominated by species that feed on aphids during their larvae stages. Because aphids predominantly occur in agricultural land, reproduction of this hoverfly group was dependent on the presence of managed fields. This result underlines that source-sink dynamics of FVIs in the agricultural landscape are highly dependent on the FVI species/group of concern. For predictions of these dynamics it is highly important to have sufficient information with respect to habitat requirements and life cycle of the species of concern. Further ecological studies characterizing ecological traits are particularly needed in FVI groups other than bees (section 2.3). For example, as outlined in section 7.2.4 individual-level behaviour in terms of dispersal, reproduction, mortality and individual/environmental interactions might be important parameters.

In short: Source-sink dynamics and risk mitigation measures

The long-term persistence of FVI populations in agricultural landscapes depends on a balance between population sources (e.g. semi-natural habitats) and population sinks (e.g. agricultural fields, pesticide influenced off-field habitats). In general, the persistence of FVI populations is promoted by strengthen the sources and/or reducing population sinks.

Possibilities to strengthen population sources:

- Preservation of large off-field habitats (e.g. grasslands, forest, nature reserves) in agricultural landscapes.
- Management of existing off-field habitats to increase pollen resources.
- Achieving a heterogeneous landscape by implementing linear structured mitigation measures (e.g. wildflower in-field buffer strips) and areal structured mitigation measures (e.g. fallows).

Possibilities to reducing population sinks:

- Reduction of pesticide applications (e.g. rate, intervals, frequency)
- Use of spray drift reducing techniques
- Timing of the application
- Management of vegetation in sink habitats with respect to improve the quality of existing habitats and to create new habitats in order to provide resources for growth, maintenance and reproduction of FVI populations.

However, certain FVI groups show a reversed pattern of the normally expected source-sink relationship, i.e. (semi-)natural habitat acting as a source and fields acting as a sink. Hence, it is important to have sufficient information with respect to habitat requirements and life cycle of the species of concern, to adequately predict the dynamics and the consequences of risk mitigation measures.

8.4 Ecological focus areas

8.4.1 Overview

The EU common agricultural policy (CAP) was reformed in 2013 amongst others with the aim to strengthen a sustainable, ecological agriculture through a “Greening” component of direct payments (EU 2013). Besides the measures crop diversification and maintaining existing permanent grasslands the “Greening” includes the designation of ecological focus areas (EFAs) with the aim to maintain biodiversity and natural resources (see also section 8.5.2) (EU 2013; BMEL 2015b). According to Regulation (EU) No 1307/2013 (EU 2013) EFAs are defined either as areas without agricultural production or as productive areas with reduced pesticide inputs[7] (Table 48). In 2014, EU member states had to define which type of EFAs proposed by the EU is applicable in the respective member state. In Germany, all EFAs proposed by Reg (EU) No 1307/2013 are adopted and considered as EFAs, except for the type: hectares of agro-forestry (Table 48). In general farmers with more than 15 ha arable land, are obliged to designate at least 5% of their arable land as EFA in order to obtain direct payments through the 1st pillar of the EU Common Agricultural Policy. The main aim of the EFAs is to generally improve biodiversity in agricultural landscapes and to maintain natural resources.

In the following sections, the different types of EFAs (listed in Table 48) are evaluated with respect to their potential to reduce pesticide entries in adjacent off-field habitats and to promote abundance and diversity of FVI communities in agricultural landscapes (8.4.2). Moreover, several aspects which are considered as important factors when implementing landscape-related measures such as EFAs are discussed:

- Feasibility and acceptability of the implementation of EFAs by farmers (section 8.4.3).

[7] In June 2017 it was decided on EU-level that the use of pesticides should be banned in all types of ecological focus areas.

- Minimum requirements of EFAs (e.g. size, number of areas, life-time) which might be necessary to efficiently compensate population relevant effects on FVI communities as a side-effect caused by the usage of pesticides in agricultural landscapes (section 8.4.4).
- Should EFAs considered as biotopes which deserving protection? Are additional legal requirements necessary to protect ecological focus areas against pesticide entries (section 8.4.5)?

Table 48: Types of ecological focus areas implemented in Germany according to Regulation (EU) No 1307/2013 and their potential to reduce pesticide inputs in off-field habitats and to promote FVIs in agricultural landscapes.

	Type of ecological focus area	Efficiency on reduction of pesticide inputs in adjacent off-field habitats	Efficiency on the increase of abundance and diversity of FVI
Without agricultural production	**Land lying fallows**	x	x
	Terraces	-	o
	Landscape features		
	Hedges or wooded strips	x	x
	Isolated trees	(x) [a]	o
	Trees in a line	(x) [a]	o
	Trees in a group	(x) [a]	o
	Field margins	x	x
	Traditional stone walls	(x)[b]	o
	Ponds and ditches (riparian veget. incl.)	(x)[c]	o
	Buffer strips	x	x[d]
	Strips of eligible hectares along forest edges	x	x
Productive areas	**Areas with short rotation coppice with no use of mineral fertiliser and/or pesticides**	x	o
	Afforested areas	(x)[e]	o
	Areas with catch crops, or green cover	-	o
	Areas with nitrogen-fixing crops	-	o

x: efficiency scientifically demonstrated; (x) efficiency scientifically demonstrated with limitations; (o) insufficient database, but efficiency assumed based on ecological considerations, (-) not/minor efficient

[a] depending on understorey.

[b] in case of very dense, non-porous barriers (as in case of stone walls) there is a wall effect, the spray cloud climbs over the wall and the filter efficiency decreases. Therefore the efficiency to reduce pesticide spray drift entries in adjacent non-target habitat is restricted on the area directly behind the wall and might be lower than in case of high vegetation.

[c] depending on location and available riparian vegetation (height, density).

[d] strips seeded with wildflowers (or with nectar and pollen mixture) were most effective in increasing abundance and species richness of bees, butterflies and hoverflies. Strips with natural vegetation seem to be partly less effective for bee and butterfly populations.

[e] depends on the amount of pesticides which are used in afforested areas. Usually the amount of applied pesticides is much lower in forestry compared to agricultural fields.

Definitions of ecological focus areas which can be declared in Germany

In the following, definitions of EFAs are shortly summarized with respect to minimum size, location and other type-specific characteristics according to Commission Delegated Regulation (EU) No 639/2014 (EC 2014). Additional requirements in Germany are pointed out according to "Bundesministerium für Ernährung und Landwirtschaft" (BMEL 2015b):

Land lying fallows: Agricultural fields which are temporarily taken out of production. The funding as ecological focus area requires that fallows are established for at least six months .In Germany, a minimum area of 0.03 to 0.3 ha[8] is required and the application of pesticides on these areas is not allowed. Fallows can be left to the natural regeneration process or seed mixtures[9] can be seeded.

Terraces: Terraces are defined as linear and vertical levelled surface structures and are primarily established to decrease the slope of acreages for suitable agricultural use of hillslopes by farmers.

Landscape features: Including **hedges** or **wooded strips**, **isolated trees**, **trees in a line** or **in a group**, **field margins, traditional stone walls**, **ponds** and **ditches**. Already existing landscape features (e.g. single tree (natural monument) in a crop) can also be declared as EFA if they are located within or adjacent to an arable land.

- *Hedges or wooded strips:* Structures adjoining to agricultural sites with maximum mean width of 10 m. In Germany a minimum length of 10 m is mandatory and the required maximum width is 15 m.
- *Isolated trees:* Freestanding trees with a crown diameter of minimum 4 m.
- *Trees in a line:* A number of trees in line with a crown diameter of minimum 4 m and a space between the crowns of maximum 5 m. In Germany trees in a line have to consist of minimum five trees in a row with a minimum length of 50 m.
- *Trees in a group:* A group of trees which are connected by their crowns and field copses with an area of maximum 0.3 ha. In Germany a total area of 0.005 to 0.2 ha can be declared as EFA.
- *Field margins:* Field margins declared as EFA are defined as permanently present off-field strips inside or between agricultural acreages with a width of 1-20 m. There is no agricultural production and therefore no pesticides are allowed. In Germany, field margins are usually overgrown with grassy and herbaceous plants (also higher vegetation like hedges or trees can be present).
- *Traditional stone walls*: EU member states are obliged to set requirements with regard to the minimum size. In Germany, traditional stone walls can be defined either as walls consisting of soil and clay or as heaped natural stones with a minimum length of 5 m, respectively.
- *Ponds and ditches:* Ponds are generally defined as surface water up to a maximum area of 0.1 ha. Concrete or plastic shells used as pond reservoir are not sufficient to be classified as EFA. EU Member States can decide, if a riparian vegetated strip around the pond with a width up to 10 m should be part of the declared EFA. However, this possibility is currently not implemented in Germany. Furthermore, in Germany for ponds and similar wetlands (e.g. dolines and kettle holes) the maximum area is limited to 0.2 ha. Ditches can be declared with a maximum width of 6 meters. The EU regulation further specifies that open watercourses can also be declared as ditch. However, it is currently not clear (based on the EU and the German regulation) if ditches should have water permanently or if also periodically water bodies are included. Ditches with walls of concrete are not acceptable to be classified as EFA.

Buffer strips: Buffer strips (including strips which are parallel adjacent to water bodies) with minimum width of 1 m and where no pesticides are applied can be declared as EFA. In Germany, strips are differentiated in field edges and buffer strips. Field edges are strips at the edge of field or between two fields (natural regenerated or sown with seed mixtures [9]) where no pesticides are applied with a width of 1 to 20 meters. Field edges located along watercourses are defined as buffer strips. Buffer strips must meet the same requirements as field edges. Buffer strips may include riparian vegetation up to a width

[8] Required size depends on the respective German federal state.

9 It is dependent on the respective federal state which mixture is allowed to be seeded. For example, in North Rhine-Westphalia it is possible to seed wildflowers, grass mixtures and herbaceous fodder plants Landwirtschaftskammer NRW (2017): Wegweiser Biodiversität in der Landwirtschaft Maßnahmenblatt: Ökologische Vorrangflächen im Greening. 3.

of 10 m.
Strips of eligible hectares along forest edges: Buffer strips with a width of 1-10 m located directly between arable land (e.g. crop, fallow) and forest can be declared as the EFA type "strips of eligible hectares along forest edges". In Germany on these strips, there is no agricultural production allowed and pesticides are not applied. In general, strips along forest edges can be left to the natural regeneration process or seed mixtures [9] can be seeded.
Areas with short rotation coppice with no use of mineral fertiliser and/or pesticides: Short rotation coppices with no use of mineral fertilizer and/or pesticides can be funded as EFA. In Germany, areas where special tree species (e.g. *Salix viminalis*, *Populus alba* or *Quercus robur*) are planted on a minimum area of 0.03 to 0.3 ha [8] can be declared as short rotation coppice. These areas are defined as "permanent crops" with an activity time of three to ten years. In addition in Germany, the use of pesticides and fertilizer is only prohibited in the claim year.
Afforested areas: Afforested or reforested areas (forest or woodlands) created in agricultural landscapes owned by the farmer can be declared as EFA. In Germany, a minimum area of 0.03 to 0.3 ha [8] is required.
Areas with catch crops, or green cover: Areas with catch crops are defined as areas sown with mixtures of different cultivated plants between successive plantings of main crops. Areas with green cover are defined as grasses which are sown under main crops. In Germany, for catch crops plant species such as *Sinapis arvensis*, *Raphanus sativus* or *Trifolium pratense* in an area with a minimum size of 0.03 to 0.3 ha [8] can be declared as EFA. Furthermore, the application of pesticides is only prohibited in the claim year after the harvest of the main culture. In areas with green cover, all grassy seeds are allowed. However, sowing of clover mixtures is not permitted to be sown under the main crop.
Areas with nitrogen-fixing crops (legumes): Nitrogen-fixing-crops are defined as legume crops which are at least present during one growing season. In Germany legumes such as *Glycine, Lupinus albus* or *Trifolium pratense* sown either as pure culture or as mixed culture on an area of minimum 0.03 to 0.3 ha [8] can be declared as EFA. It is additionally required that the area is exclusively sown with legume species.

8.4.2 Evaluation of different types of ecological focus areas

8.4.2.1 Land lying fallow

Land lying fallow (arable fallow) are defined as agricultural fields which are temporarily taken out of production (EC 2014; Toivonen et al. 2016). The funding as ecological focus area requires that fallows are established for at least six months (Council of the European Union 2016). Moreover, in Germany a minimum area of 0.03 to 0.3 ha (depending on respective federal state) is mandatory and the application of pesticides on these areas is not allowed (BMEL 2015b; Nitsch et al. 2016). In general, arable fallows can be left to the natural regeneration process or seed mixtures can be seeded (BMEL 2015b). It is dependent on the respective federal state which mixture is allowed to be seeded. For example, in North Rhine-Westphalia it is possible to seed wildflowers, grass mixtures and herbaceous fodder plants (Landwirtschaftskammer NRW 2017).

Land lying fallows can reduce pesticide entries in non-target terrestrial off-field habitats and FVI might benefit from the presence of land lying fallows in in the agricultural landscapes. For further information, please refer to section 8.2.3.

8.4.2.2 Terraces

Terraced agricultural land can be declared as EFA (EC 2014). Terraces are defined as linear and vertical levelled surface structures and are primarily established to decrease the slope of acreage for a suitable agricultural use of hillslopes by farmers (Bevan & Canolly 2011; BMEL 2015b).

We do not assume that terraces contribute to a reduction of pesticide entries in non-target terrestrial habitats. There were not any indications in literature. However, terraces in agricultural landscapes might have positive impacts on FVIs such as cavity nesting bees so called renter (please refer to section 5.1.1.3) by providing habitats and nesting places in case that the step structure consists of stones (e.g. dry stone walls or gabions) (Figure 31). However, studies for evaluating the extent of this effect on FVI populations are currently not available. Thus, further research is necessary for a final assessment of terraces with respect to FVIs. Nevertheless, based on ecological considerations (as outlined above) efficiency on FVIs by terraces can be assumed.

Figure 31: Typical structure of a terrace in agricultural landscapes

Source: own picture, ecotox consult

8.4.2.3 High vegetation such as hedges, shrubberies, trees

Agricultural landscape features such as hedges, shrubberies or trees can be declared as ecological focus area when minimum requirements are met. Hedges or wooded strips adjoining to agricultural sites up to a width of 10 meters, trees in a line or trees which are isolated with a diameter of minimum 4 meters and trees in groups with a maximum area of 0.3 ha can be funded as ecological focus area (EC 2014). In addition, in Germany for hedges a minimum length of 10 m is mandatory and the required maximum width is 15 m. Furthermore, trees in a line have to consist of minimum five trees in a row with a minimum length of 50 m. Trees in a group with a total area of 0.005 to 0.2 ha can also be declared as EFA in Germany (BMEL 2015b).

High vegetation (particularly hedges) can reduce pesticide entries in off-field habitats behind the vegetation barrier. In addition, FVI might benefit from the presence of high vegetation in the agricultural landscape. Further details can be found in section 8.2.4.

8.4.2.4 Traditional stone walls

EU member states are obliged to set requirements for traditional stone walls with regard to the minimum size (EC 2014). In Germany, stone walls can be defined as stone walls with a minimum width of 5 m consisting either of soil and clay or of heaped natural stones, respectively (BMEL 2015b). Stone walls can act as barrier within agricultural crops. Therefore, they might reduce pesticide spray drift entries into non-target habitats located directly behind the wall. As outlined for high vegetation the extent of pesticide exposure reduction is dependent on the height of the barrier. However, in comparison to hedges, stone walls might have a lower filter effect, and consequently the percent pesticide reduction might be lower. In case of very dense, non-porous barriers (as in case of stone walls) there is a wall effect which drives the spray cloud to climb over the wall and the filter efficiency decreases (Ucar & Hall 2001; Koch et al. 2003). In addition, studies focusing on the drift reducing effect of stone walls in agricultural landscapes are currently not available in literature.

Several authors recommend traditional stone walls to enhance the habitat availability for several wild bees (Jahn et al. 2014; Gottwald & Stein-Bachinger 2016; Plattform Bienenzukunft 2016). Piled big rock fragments are used by bees such as cavity nesting bee species (e.g. *Bombus terrestris or Osmia bicornis*) so called renter (please refer to section 0) nesting habitat. Gottwald & Stein-Bachinger (2016) reported that particularly mason bees, furrow bees or leaf cutter bees might benefit from traditional stone walls. Furthermore, specifically for bumble bees the creation of dry stone walls may provide many habitats in available cavities (Jahn et al. 2014). In addition, Plattform Bienenzukunft (2016) recommends the filling of gaps inside the wall with sand which might optimize the conditions for FVI nests. However, the scientific databases of this recommendation are not clear. Nevertheless, based on ecological considerations, efficiency on FVIs by traditional stone walls can be assumed.

8.4.2.5 Field margins

Field margins declared as ecological focus area are defined as permanently present off-field strips inside or between agricultural acreages (BMEL 2015b) with a width of 1 m to 20 m (EC 2014). There is no agricultural production and therefore no pesticides are allowed. In Germany, field margins are usually overgrown with grassy, herbaceous vegetation or higher vegetation such as hedges or trees (BMEL 2015b).

Depending on their width and present vegetation type, field margins can efficiently reduce pesticide entries in adjacent off-field habitat (for detailed discussion, please refer to section 8.2.2.1). Moreover, FVIs can particularly benefit from permanently present field margins (see section 8.2.2.2).

8.4.2.6 Ponds and ditches

General Information

Landscape features such as ponds and ditches inside or between agricultural acreages which are owned by farmers can be declared as EFA. In the context of EFAs ponds are defined as surface water up to a maximum surface area of 0.1 ha (EC 2014). Concrete or plastic shells used as pond reservoir are not sufficient to be classified as EFA. EU member states can decide if a riparian vegetated strip around the pond with a width up to 10 m should be part of the declared EFA. However, this possibility is currently not implemented in Germany. Furthermore, in Germany for ponds and similar wetlands (e.g. dolines and kettle holes) a maximum area of 0.2 ha is set (BMEL 2015b). Ditches can be declared with a maximum width of 6 meters (EC 2014). The EU regulation further specifies that open water courses can also be declared as ditch. Ditches with walls of concrete are not acceptable to be classified as EFA. However, a further characterisation of ditches based on European regulation (EC 2014) and German regulation (BMEL 2015b) is currently not available. There is no information e.g. with regard to the requirements of aspects such as lentic or lotic systems, having permanently or non-permanently water during the course of the year.

Reduction of pesticide entries

Adjacent to arable land ditches and their riparian strips increase the distance between the location of pesticide application (the field) and other terrestrial off-field habitats. In case that high riparian vegetation is present the reduction of pesticide entries into other, more distant off-field habitats is further promoted (please refer to section 8.2.4.1). At this point it is important to note that certainly there might be reductions of pesticide entries in off-field habitats, but pesticides might then probably reach the ditch and its riparian vegetation. However, farmers in Germany must comply with distance requirements when applying pesticides adjacent to waterbodies (e.g. as a risk mitigation measure applied in the pesticide risk authorization process for a specific pesticide use). Furthermore, in some federal states in Germany (e.g. Lower Saxony, North Rhine-Westphalia or Saxony-Anhalt) there are minimum distances of 1 m to a waterbody generally required. Dependent on the width of the required non-spray zone, pesticide entries in ditches might be lowered.

Effects on FVI

FVI populations might benefit from the presence of ponds and ditches in the agricultural landscape (Diekotter et al. 2006; Kleijn & van Langevelde 2006; Stewart et al. 2017). The main results of these studies are presented in Table 49. These waterbodies can provide important foraging resources or habitat and nesting possibilities for several riparian specialized FVI species which is for example shown by Diekotter et al. (2006) for the bumblebee *B. muscorum*.

During a field-study in Marburg (Germany), the authors investigated effects of ditches (among other landscape features) on the bumble bee species *B. muscorum*. The abundance of these species was positively significantly affected by the presence of ditches. Furthermore, the authors assumed that ditches (and also brooks) might represent an essential habitat for the bee species *B. muscorum.* This assumption by Diekotter et al. (2006) is based on the positive significant effect of ditches on *B. muscorum*, the nest-searching behaviour of queens and the two observed bee nests along in-field ditches and brooks.

Kleijn & van Langevelde (2006) investigated Dutch ditches (average width of 2.53 m) in semi-natural landscapes and their influence on FVI (bees and hoverflies) species richness and abundance within a radius of 250-2000m around the ditches. Semi-natural habitats were defined as forests, heather, cemeteries, swamps and reed beds. Ditches at any length or the number of inflorescences in semi-natural habitats had no significant effects on hoverfly species richness and abundance in any landscape radius. In contrast, inflorescences in a 500 m radius around the ditch had positive significant effects on wild bee (99% of wild bees were bumble bee species) species richness and abundance. But the length of ditch had also neither significant effect on bee species richness nor on bee abundance. Thus, the positive effect on bee species richness and abundance was linked by the authors to the number of inflorescences not to the available ditches. It is important to note, that this study exclusively focused on ditches within semi-natural habitats which might be also highly attractive for FVI. In case that ditches are located in agricultural land, the results of such an investigation could be completely different, particularly when other attractive habitats are rare.

Stewart et al. (2017) investigated effects of semi-natural vegetation (dominated by plants, trees and shrubberies) in agricultural landscapes with and without ponds (volumes between 3500 and 8000 m^3

and diameters between 20 and 70 m) on abundances of wild bees and hoverflies in comparison to a control habitat (cropland dominated by cereals and oil seed rape) during a study in Sweden. Significantly higher abundances of hoverflies and wild bees were found in semi-natural vegetation habitats including ponds than in the control habitat. Also a significant positive effect on hoverfly abundance was observed in semi-natural habitats with ponds compared to the pondless habitat. However, no significant difference in wild bee abundance was observed between semi-natural habitats with and without ponds. In conclusion, the authors presumed that ponds might have the potential to increase landscape heterogeneity as well as complexity and constitute an attractive habitat for FVI communities, particularly for hoverfly species.

In conclusion, there are indications that ditches or ponds might be suitable landscape features with respect to the protection of different FVI groups in agricultural landscapes (Diekotter et al. 2006; Stewart et al. 2017). Based on the results of a literature review Sumpich (2011) assumed that particularly Lepidoptera species could benefit from stream banks or wetlands since butterflies and moths are closely associated (e.g. inhabiting there) with such places close to water. Water plants in agricultural water bodies, such as water lily and reed, can provide important nesting and foraging habitats for bees and wasps as shown by Heneberg et al. (2014). Furthermore, the presence of surface waters in agricultural landscapes might positively influence plant species richness in the vicinity (van Dijk et al. 2014), which is considered to be an important influencing factor for the increase of abundance and species richness of FVI communities (please refer to section 8.2.3.2). Because, the current database is rather small further research is necessary to assess the effect of ditches and ponds on FVI communities in more detail. However, based on ecological consideration, efficiency on FVIs by ponds and ditches can be assumed.

Table 49: Effects of ponds and ditches on flower-visiting insects (FVI) as investigated in field studies (n.r. = not reported). “Yes” means that a statistically significant effect was observed, unless otherwise is stated.

			Positive effect on FVI				
Reference	Ponds/ ditches	Size of pond/ditch	Abundance	Species richness	Investi-gated taxa	Adjacent crop	Monitoring period
Diekotter et al. (2006)	Ditches	n.r.	**Yes**	Not relevant	*B. muscorum*	Agricultural field	2 years
Klein & van Langevelde (2006)	Ditches	2.53 m wide (average)	No	No	Wild bees, Hoverflies	-[a]	1 year
Stewart et al. (2007)	Ponds	Volumes between 3500 and 8000 m^3 and diameters between 20 and 70 m	**Yes**[b]	n.r.	Wild bees, Hoverflies	Cereals and oil seed rape	1 year

[a] The study exclusively focused on ditches within semi-natural habitats. However, ditches within arable land were not investigated. Therefore, no adjacent crop is available.
[b] The authors compared vegetated pond habitat with control habitat (crop) and vegetated pondless habitat. Only the comparison of pond and pondless vegetated habitat showed no significant difference in wild bee abundance.

8.4.2.7 Buffer strips

Buffer strips (including strips which are parallel adjacent to water bodies) with minimum width of 1 m and where no pesticides are applied can be declared as EFA (EC 2014). In Germany, strips are differentiated in field edges and buffer strips. Field edges are defined as strips established on the arable land at the edge of field or between two fields with a width of 1 to 20 m. Agricultural production on these strips is not allowed and consequently no pesticides on these strips are applied. Strips located along water courses are defined as buffer strips. In general, field edges can be left to the natural regeneration process or seed mixtures can be seeded (BMEL 2015b). It is dependent on the respective federal state which mixture is allowed to be seeded. For example, in North Rhine-Westphalia it is possible to seed wildflowers, grass mixtures and herbaceous fodder plants (Landwirtschaftskammer NRW 2017). Buffer strips must meet the same requirements as field edges. Buffer strips may include riparian vegetation up to a width of 10 m (BMEL 2015b).

Depending on their width and present vegetation field edges and buffer strips can efficiently reduce pesticide spray drifts entries in adjacent non-target terrestrial habitats and to positively affect FVIs in the agricultural landscape. Detailed information regarding this topic can be found in section 8.2.

8.4.2.8 Strips of eligible hectares along forest edges

General Information

Buffer strips with a width of 1-10 m which are located directly between arable land (e.g. crop, fallow) and forest can be declared as the EFA type "strips of eligible hectares along forest edges" (EC 2014; BMEL 2015b). In Germany, on these strips, there is no agricultural production allowed and pesticides are not applied. In general, strips along forest edges can be left to the natural regeneration process or seed mixtures can be seeded (BMEL 2015b). It is dependent on the respective federal state which mixture is allowed to be seeded. For example, in North Rhine-Westphalia it is possible to seed wildflowers, grass mixtures and herbaceous fodder plants (Landwirtschaftskammer NRW 2017).

Reduction of pesticide entries

The strips might act as buffer between off-field habitats and arable land and consequently pesticide drift entries in adjacent off-field habitats (i.e. the forest) are reduced. With regard to the efficiency of reducing pesticide spray drift entries, please refer to section 8.2.

Effects on FVI

Strips of eligible hectares along forest edges might support FVI communities by enhancing their diversity and abundance as already shown for in-field buffer strips (8.2). In particular, FVI species which might prefer the presence of surrounding forests might additionally benefit from strips created near to the forest edge. There are few studies describing effects of strips along forest edges on FVIs in agricultural landscapes.

Korpela et al. (2013) investigated in a 4-year study the effects of wildflower strips located at different positions in the agricultural landscape (i.e., next to forest, strips at the middle of a reed canary grass field plot, strips adjacent to crop) on bumble bees, butterflies and diurnal moths. The strips (consisting of *Centaurea jacea L.*, *C. phrygia L.*, *Leucanthemum vulgare Lam.*, *Trifolium repens L.* and *Agrostis* capillaris *L.*) were established on a reed canary grass plot between forest and cereal crop. The authors observed no difference in species diversity between the three different treatments. However, the authors showed that bumble bee abundance was significantly higher in wildflower strips located in the middle of the reed canary grass field than in strips close to the crop (spring cereal). Korpela et al. (2013) presume that the high *Centaurea* flower coverage in the centre strips might have positively influenced the bumble bee abundance. Furthermore, a positive significant effect on butterfly abundance in the strip next to forest was observed. The authors presume a positive relationship between butterfly abundance and the adjacent forest. Based on an analysis of the same data as used from Korpela et al. (2013), Miettinen et al. (2014) showed positive effects of 5-m wide sown biodiversity strips between a barley-field and a forest on bumble bees, diurnal moths and butterflies compared to a feed-barley field (conventional production) directly adjacent to the forest.

These findings indicate that strips of eligible hectares between arable land and forest might have the potential to improve FVI communities. As shown for in-field buffer strips (please refer to section 8.2) different FVI species might be attracted by strips which are seeded with wildflower mixtures or left to the natural regeneration process. The data of Korpela et al. (2013) indicate that particularly Lepidoptera species might benefit from strips of eligible hectares along forest edges. But also bumble bees were attracted by these strips (Miettinen et al. 2014). Furthermore, it could be assumed that forest edges might be an important factor with respect to protection of wild bee species (e.g. *Lasioglossum malachurum, Hylaeus communis or Osmia bicornis*) which prefer to use forest edges as habitat (please refer to section 3.2.1).

With one study investigating the impact of wildflower strips adjacent to forest edges, the available scientific data base should be considered as insufficient. However, there are numerous studies investigating the effect of in-field buffer strips but not adjacent to a forest (please refer to section 8.2). These studies reveal that in-field wildflower strips (or with nectar and pollen mixture) are most effective in increasing abundance and species richness of bees, butterflies and hoverflies. Buffer strips with natural vegetation or grassed buffer strips seem to be partly less effective. Based on these data we assume that also strips of eligible hectares along forest edges might positively affect FVIs, at least if wildflower mixtures are seeded.

8.4.2.9 Areas with short rotation coppice with no use of mineral fertilizer and/or pesticides

General Information

Areas with short rotation coppice with no use of mineral fertilizer and/or pesticides can be funded as EFA. EU member states are obliged to set additional requirements with regard to the management and minimum size (EC 2014). In Germany, areas where special tree species (e.g. *Salix viminalis*, *Populus alba* or *Quercus robur*) are planted on a minimum area of 0.03 to 0.3 ha (depending on the respective federal state) can be declared as short rotation coppice (BMEL 2015b; Nitsch et al. 2016; Rueb 2016). These areas are defined as “permanent crops” with an activity time of three to ten years (EU 2013; Nitsch et al. 2016). In addition the use of pesticides and fertilizer is only prohibited in the claim year in Germany (BMEL 2015b).

Reduction of pesticide entries

Areas with short rotation coppice with no use of mineral fertilizer and/or pesticides might have the potential to reduce amounts of pesticide entries in off-field habitats. EU member states have to decide the legal requirements of areas with short rotation coppice with respect to the use of chemicals (EC 2014). In Germany, the use of pesticides and mineral fertilizer is at least not allowed in the claim year. Thus, pesticide entries might be reduced in this period. Furthermore, the amount of pesticides applied in areas with short rotation coppice is lower in comparison to the amount which is applied in conventional agricultural crops (Ledin 1998; Perttu 1998; EC 2014).

Effects on FVI

FVI populations might benefit from areas with short rotation coppice. For example, by implementation of short rotation coppices new ecological nesting structures in agricultural landscapes for FVI species are created. According to Nitsch et al. (2016), particularly, by a section-by-section harvesting of the coppice or the combination of different tree species the diversity with respect to the structure of the habitat and thus, the positive effects on FVIs can be additionally increased. However, the scientific database of this recommendation is not clear.

During a German field study, Haß et al. (2012) investigated influences of a short rotation coppice (area of 8 ha with willow plantations: *Salix*) on *Apis mellifera* and wild bees in comparison to a fallow, a field boundary, a boundary between the willow rows, and a path. Most bee species were observed on fallows, closely followed by the short rotation coppice. The authors showed a positive relationship between the different flowering time of willows or dandelions (which were present on fallows) and bee density on the area with short rotation coppice. However, no statistical analysis was conducted. In April bees were only observed on willow plants, in May bees were only observed on dandelions in the fallow. The authors presume that areas with short rotation coppice might provide nesting and foraging habitat for bee communities, particularly early-year bees like bumble bees or others which are specialised on *Salix* trees. However, the authors pointed out that short rotation coppices only have short flowering times and therefore the presence of e.g. semi-natural biotopes are nevertheless important.

Reddersen (2001) investigated the influence of short rotation coppice i.e., willow plantations (*Salix viminalis* trees) on the occurrence of bumble bees and honey bees. The plantations (1.1 ha each plantation) were established adjacent to an intensive Danish farmland and investigated for three years. In this study only few bees were observed in the investigated plots and no correlation between bees and short rotation coppices could be observed. The authors assumed, that the availability of pollen and nectar were insufficient for an attraction of bumble bees and honey bees by willow plantations. Furthermore, the authors supposed, when changing certain characteristics at the short rotation coppice area (e.g. plant species, age and sex of plant species) the importance for FVI would rise as well. To achieve the recommended modification, Reddersen (2001) suggests for example establishing willow species with different flowering times, subdividing coppice plots into subplots with different harvest cycles or planting both male and female plants (male produce both nectar and pollen while females can produce seeds when they are pollinated).

In conclusion the database with regard to effects of short rotation coppices on FVI is pretty small (i.e. only two available studies) (Reddersen 2001; Haß et al. 2012). Only Haß et al. 2012 could show positive influences of areas with short rotation coppice on FVIs by providing nesting and foraging habitats for FVI communities. However, in this study no statistical analysis was conducted. Reddersen (2001) could not show positive effects, but assumes that an optimization of short rotation coppices with regard to planted species or harvesting cycles might enhance the importance of such biotopes for FVIs. Furthermore, the presence of additional biotopes in the agricultural landscape and the connection to short rotation coppices might be useful to provide a continuous nectar and pollen supply for FVIs. The flowering time of trees in short rotation coppices is actually very short. With regard to short rotation coppices further research on the effects on FVI in agricultural landscape is needed. However, based on ecological consideration, efficiency on FVIs by areas with short rotation coppice can be assumed.

8.4.2.10 Afforested areas

General Information

Afforested or reforested areas (forest or woodlands) created in agricultural landscapes owned by the farmer can be declared as EFA (EC 2014). In Germany a minimum size of 0.03 to 0.3 ha (depending on the respective German federal state) is required for afforested areas (BMEL 2015b; Nitsch et al. 2016; Rueb 2016).

Reduction of pesticide entries

In general, the amount of applied pesticides (mostly herbicides, but also insecticides and fungicides) are much lower in forestry compared to agricultural fields (usually less than 1 % of the amount in agricultural areas) throughout in Europe (Willoughby et al. 2009). Consequently, FVIs in afforested or reforested areas are much less exposed to pesticides than in cultivated crops. Furthermore, off-field habitats adjacent to afforested/reforested areas receive consequently lower pesticide entries (e.g. via spray drift) than off-field habitats neighbouring cultivated crops. However, for example in Germany, farmers tend to use pesticides in afforested areas more frequently when alternative mechanical controls require high effort and costs (Willoughby et al. 2009).

Effects on FVI

The large number of trees in afforested areas might provide habitat opportunities for FVIs. Particularly FVIs which prefer to visit and inhabit forest edges (please refer to section 0) could benefit from afforested areas.

Fuller et al. (2014) investigated in an 8-year study effects of afforestation on hoverflies in agricultural grasslands in Ireland. Five study sites were analysed 1-year before the planting of various tree species (e.g. maple (*Acer pseudoplatanus L.*), larch (*Larix kaempferi*) and Sitka spruce (*Picea sitchensis Carrière*)) and 7-years after the planting. The afforestation had significant positive effects on most habitat characteristics which were assumed to be important for hoverflies (lower and upper field layer vegetation, litter and deadwood) but negative significant effects on ground vegetation. However, significant effects on species richness of hoverflies (sub-groups of open, water- and woody vegetation-associated species) could not be found. The study of Fuller et al. (2014) focused only on afforested areas established within a grassland area.

In conclusion only one available study investigated the impact of afforested areas on FVIs, i.e. hoverflies (Fuller et al. 2014). Although the habitat characteristics indicated improved habitat conditions for hoverflies, they could not show any positive effects on hoverflies by afforested areas established in landscape dominated by grasslands. However, the success of an afforested area to support general biodiversity depends on previous land use (intensively used agricultural area or natural grassland), the method of establishment and the surrounding landscape (Zanchi et al. 2007). We assume that in case that an agricultural field is located in a forested landscape, the creation of an additional forest area would not greatly promote the FVI community. In contrast, in an agricultural landscape dominated by open structures, the creation of a forest area leads to a more heterogeneous, more diverse landscape and might therefore also promote the diversity of FVIs. Furthermore, the implementation of additional management measures in these areas might additionally enhance biodiversity and thus, FVI diversity might benefit from these measures; e.g. planting of diverse tree mixtures with different growing patterns, creation of natural landscape elements (e.g. shrubberies, deadwood, stone structures or waterbodies) inside the forest and leaving some parts of the afforested area open within the forest area (permanently or for spontaneously regeneration). Based on these ecological considerations, efficiency on FVIs by afforested areas can be assumed. However, further research on the effects of afforested areas established in agricultural landscapes on FVI communities is needed.

8.4.2.11 Areas with catch crops or green cover

General Information

In the context of EFAs catch crops are defined as areas sown with mixtures of crop species between successive plantings of main crops (EC 2014). Areas with green cover (cover crops) are defined as grasses which are sown under main crops. EU member states are obliged to set additional requirements with regard to the management (e.g. which plant species are allowed) and minimum size. In Germany, for catch crops different cultivated plants species like *Sinapis arvensis*, *Raphanus sativus* or *Trifolium pratense* are available to be seeded in an area with a minimum size of 0.03-0.3 ha (depending on the respective German federal state) (BMEL 2015b; Nitsch et al. 2016; Rueb 2016). In areas with green cover, all grassy seeds are allowed. However, sowing of clover mixtures is not permitted to be sown under the main crop.

Reduction of pesticide entries

So far, in catch crops declared as EFA the application of pesticides in Germany are only prohibited in the claim year after the harvest of the main culture. In the following year pesticide applications are allowed again starting from beginning of January. Consequently, a reduction of pesticide entries in-field and via spray drift in off-field habitats is limited to a few months between midsummer/autumn and the end of a year. However, a pesticide reducing effect is only present if the catch crop replaces another winter crop where pesticides would be applied (e.g. winter wheat). In contrast to catch crops, cover crops are simultaneously planted with the main culture and pesticide applications on these areas are allowed. However, weed pressure might be lower due to undersown grasses so that lower amounts of pesticides are needed (Nicholls & Altieri 2013; Nitsch et al. 2016). Under this assumption the pesticide exposure of FVIs would be reduced in-field and consequently also in non-target off-field habitats.

Effects on FVI

Eberle et al. (2015) investigated effects of catch crops (pennycress, winter camelina, and winter canola) on various FVI groups (bumble bees, small bees, butterflies and honey bees) over 2 years in two different states in the USA. Seeds were sown into no-tillage wheat stubble (previous crop was spring wheat). In general, the three investigated catch crops had only little influence on FVIs. Bumble bees and butterflies visited the crops very scarce. A few honey bees were observed on winter canola crops during both study years. Only several small bee species were found in all three crops during the two study years, mainly in winter camelina and winter canola.

Carvell et al. (2006) investigated in a 3-year field-study in UK, effects of three different seed mixtures (consisting of *Borago officinalis*, *Raphanus sativus*, *Linum usitatissimum*, *Sinapis alba*, *Melilotus officinalis*) on bumble bee species. The seed mixtures were sown annually as catch crop and in adjacent field margins (two perennial grass and wildflower mixtures). Total bumble bee abundance was significantly higher in the catch crop compared to the field margins when considering the overall study period. Shorter-tongued species seemed to prefer the annually sown mixture in catch crop, while longer-tongued species visited rather the perennial field margins.

In general, there are indications that areas with catch crops or green cover might provide foraging resources for FVIs (Carvell et al. 2006; Lee-Mader et al. 2014; Nitsch et al. 2016). These areas may fulfil different functions such as providing foraging opportunities in late summer season (Nitsch et al. 2016; Plattform Bienenzukunft 2016) or improving the general health and reproduction potential of e.g. wild bee communities in arable lands if attractive plants are planted (Lee-Mader et al. 2014). Particularly cover crops are assumed to improve main crops by promoting additionally attractive nectar and pollen resources (Lee-Mader et al. 2014). The success of catch crops or cover crops to support FVIs might strongly be dependent on sown seed species (some flower species do not bloom anymore at late season in the year). However, even adverse effects on FVIs might occur caused by ploughing of these catch crops in late winter/early spring (i.e. nests of FVIs below and above ground might getting destroyed) and by applying pesticides early in the following year (i.e. January and February) which may negatively affect hibernating life stages of FVI in the crop as assumed by (Ehlers et al. 2014).

In conclusion only two filed studies investigating the impact of catch crops on FVIs were available in literature (Carvell et al. 2006; Eberle et al. 2015). Both studies might indicate positive effects of catch crops on certain FVI groups. However, a sound scientific evaluation of the efficiency to protect FVIs of catch crops/green covers based on these studies is not possible, due to limitations in the study design (no comparison to non-catch crop control areas). However, based on the ecological considerations described above, efficiency on FVIs by catch crops/green cover can be assumed.

Further research is necessary to assess the effect of areas with catch crops and green cover on FVI communities in more detail. Design of future field studies should not only focus on different types of areas with catch crops/green cover but should also allow the comparison between green cover crops/catch crops and controls of conventional crops (i.e. non-catch crop/non-green cover areas).

8.4.2.12 Areas with nitrogen fixing crops

Nitrogen-fixing-crops are defined as legume crops which are at least present during one growing season. Further requirements are determined by the EU member states (EC 2014). In Germany legumes such as *Glycine*, *Lupinus* albus or *Trifolium pratense* sown either as pure culture or as mixed culture on an area of minimum 0.03 to 0.3 ha (depending on respective German federal state) can be declared as EFA (BMEL 2015b; Nitsch et al. 2016; Rueb 2016). It is additionally required that the area is exclusively sown with legume species.

Reduction of pesticide entries

The application of pesticides it not prohibited on these areas (Nitsch et al. 2016). However, the presence of legumes might result in a higher pesticide and fertilizer use efficiency caused by a reduced weed competition and insect damage as shown by Cadoux et al. (2015) and the specifically nitrogen-fixing characteristic of legume plants[10] (Zahran 1999). Furthermore, Olmstead & Brummer (2008) showed in a review that adding the legumes alfalfa or red clover to corn and soybean crops may even increase the yield although pesticide use decreases. Thus, the general pesticide usage on areas planted with nitrogen-fixing crops could be assumed to be reduced compared to non-legume crops (e.g. rape seed or wheat crops). Consequently, also pesticide entries into adjacent off-field habitats might be reduced.

Effects on FVI

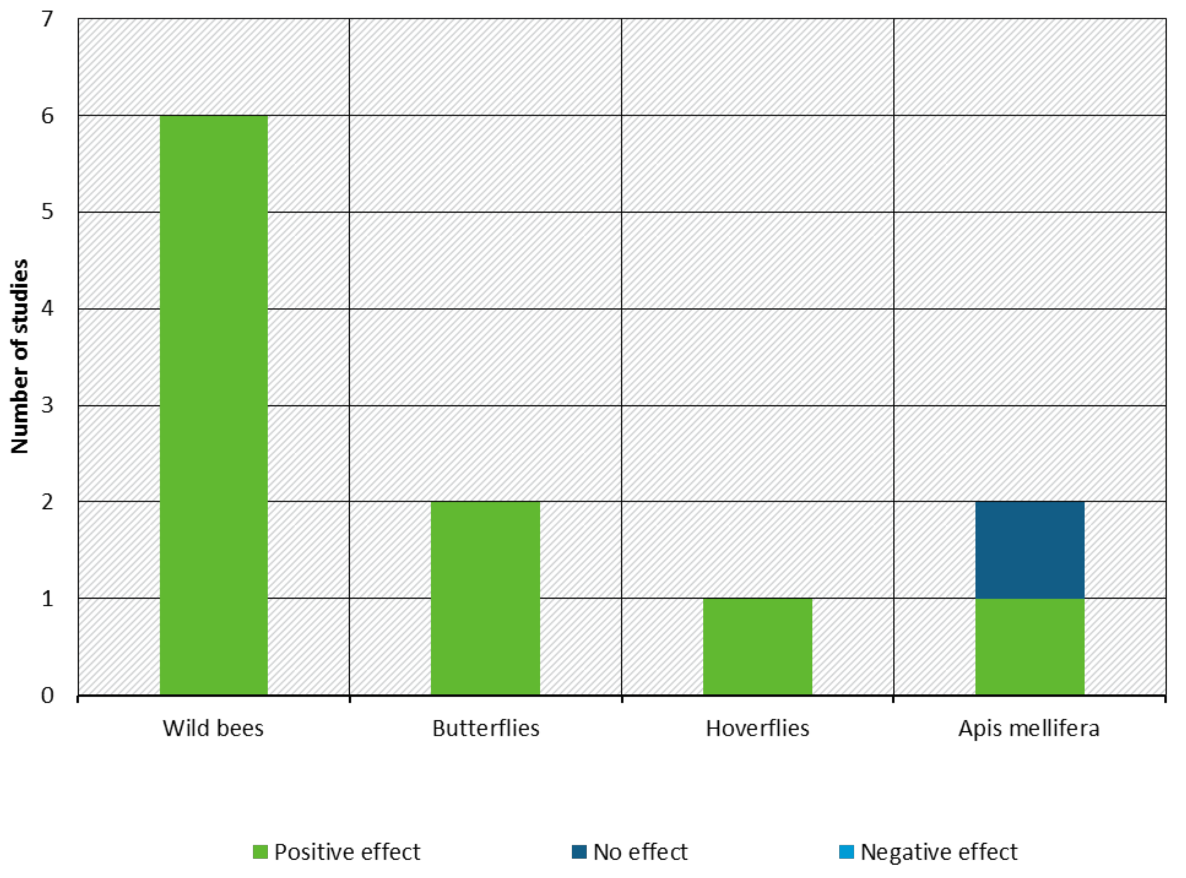
Nitrogen-fixing crops may have the potential to provide foraging and nesting resources for FVIs (BMEL 2015b) and thus, promote FVI populations. Studies focusing on areas exclusively sown with legumes and their effects on FVI are currently not available. However, numerous studies showed that flower mixtures containing amongst others legume seeds might have a positive impact on FVIs (Pywell et al. 2006; Carvell et al. 2007; Heard et al. 2007; Potts et al. 2009; Carvell et al. 2011; Woodcock et al. 2014). The main results of these studies are presented in Table 50.

The results of reviewed literature indicate that legume plants might positively affect species richness and abundance of FVIs (Table 50). Nearly all investigated seed mixtures based on legumes had positive effects on FVI species richness and abundance. Positive effects on bumble bees (Pywell et al. 2006; Carvell et al. 2007; Heard et al. 2007; Potts et al. 2009; Carvell et al. 2011; Woodcock et al. 2014), solitary bees (Woodcock et al. 2014), butterflies (Potts et al. 2009; Woodcock et al. 2014), *Apis mellifera* (Woodcock et al. 2014) and hoverflies (Woodcock et al. 2014) were observed (Table 50; Figure 32). Only in one study positive effects could not be found (Heard et al. 2007). The authors of this study could not observe any positive significant effects of legume mixtures on *Apis mellifera* sown in a forage patch compared to a typical non-crop vegetation patch. However, Woodcock et al. (2014)

[10] The root of legume plant is associated with symbiotic soil bacteria rhizobia which have the ability to fix nitrogen from air and transform it to ammonia.

showed positive significant effects of legume mixtures on *Apis mellifera,* which was even the most abundant FVI within the legume treatments. The differences in the results of both studies might be caused by the specific composition of the used legume mixtures. Woodcock et al. (2014) used mixtures containing seven different legume seeds (*Lotus corniculatus, Melilotus officinalis, Onobrychis viciifolia, Trifolium dubium, Trifolium hybridum, Trifolium pratense* and *Trifolium repens*) while Heard et al. (2007) used mixtures containing only three different legume seeds (*Trifolium pratense*, *Trifolium hybridum* and *Lotus corniculatus*). This observation indicates that the composition of seed-mixtures might be an important factor to attract certain FVI species. A study which demonstrates negative effects of seed mixtures containing amongst other legumes on FVIs could not be found.

Figure 32: Number of available studies demonstrating the effect (positive, no or negative) of seed mixtures containing amongst others legumes on different FVI taxa (bees, butterflies and hoverflies).



Source: own illustration, ecotox consult

Perennial legume species (e.g. *Medicago sativa* or *Trifolium*) or the sowing of legume-mixtures with long flowering periods might be well suited to promote FVI communities (Nitsch et al. 2016). Several authors indicated the high preference of bumble bees to visit legume mixtures containing perennial *Trifolium*, (which is additionally characterised by a long flowering period, (Pywell et al. 2006; Carvell et al. 2007; Heard et al. 2007; Potts et al. 2009; Carvell et al. 2011; Woodcock et al. 2014). In the study of Kleijn & Raemakers (2008), bumble bee species mainly collected pollen and nectar of *T. pratense*. Pollen of perennial *Lotus* species (with long flowering period) was also frequently often collected by different bumble bee species. This is in agreement with Goulson et al. (2005), who showed that especially bumble bee species might be highly attracted by legumes, particularly by *T. pratense* and *Trifolium repens* in context of pollen and nectar collecting. For further indications relating to the attractiveness of plants to certain FVI species, please refer to section 8.2.3.2.

Furthermore, the results of Carvell et al. (2011) indicated that occurrences of rare bee species like *Bombus muscorum* or *Bombus humilis* might benefit from patches sown with legume seeds. All observed rare bumble bee species only visited legume patches whereas unsown control patch were not visited by individuals of theses rare species.

In conclusion, there are strong indications that arable land sown with legume mixtures including attractive legume species (e.g. perennial, long-flowering *Trifolium*) might provide foraging and nesting resources for a wide variety of FVIs. Thus, the EFA *nitrogen-fixing crops* might be an important influencing factor for the protection of abundance and diversity of FVI communities in the agricultural landscape.

However, all studies from open literature observing positive effects of legume mixtures did not include the usage of pesticides as a factor in their study design. Hence, the direct extrapolation of these results to the EFA *nitrogen-fixing crops* – where the application of pesticides is permitted by Reg (EU) No 1307/2013 – is associated with some degree of uncertainty. In order to close this knowledge gap further research is necessary. For a general discussion on the usage of pesticides in EFAs please refer to section 8.4.5.

In short: nitrogen-fixing crops

- In general, the usage of pesticides in nitrogen-fixing crops is legally permitted.
- However, the presence of legumes might result in a higher pesticide and fertilizer use efficiency caused by a reduced weed competition and insect damage and the specifically nitrogen-fixing characteristic of legume plants. Thus, the general pesticide usage on areas planted with nitrogen-fixing crops could be assumed to be reduced compared to non-legume crops. Consequently, also pesticide entries into adjacent off-field habitats might be reduced.
- Mixtures containing legume plants might positively affect FVI abundance and/or species richness. Positive effects were shown for bumble bees, wild bees, butterflies, hoverflies *and Apis mellifera*.
- Perennial, long-flowering leguminous plants (e.g. *Trifolium pratense, Trifolium repens*) are attractive pollen and nectar resources for FVIs.

Table 50: Effects of areas sown with seed mixtures containing amongst others legumes on flower-visiting insects (FVI) as investigated in field studies (n.r. = not reported). "Yes" means that a statistically significant effect was observed, unless otherwise is stated.

Reference	Legume type		Positive effect on FVI				
		Area of sown legume	Abundance	Species richness	Investigated taxa	Adjacent crop	Monitoring period
(Carvell et al. 2007)	Pollen and nectar mixtures containing legumes.	6 m width / 50 m length	**Yes**	**Yes**	Bumble bees	Cereals	3 years
(Carvell et al. 2011)	Legume-grass mixture	0.25-1.0 ha	**Yes**	**Yes**	Bumble bees	Cereals	3 years
(Heard et al. 2007)	Legume-grass mixture	0.26 to 1 ha	**Yes**	**Yes**	Bumble bees	Several crops	2 years
(Heard et al. 2007)	Legume-grass mixture	0.26 to 1 ha	No	No	*Apis mellifera*	Several crops	2 years
(Woodcock et al. 2014)	Legume-forb-grass mixture	~ 0,09 ha	**Yes** [a]	**Yes** [a]	Wild Bees *Apis mellifera* Butterflies Hoverflies	Several crops	5 years (4 year sampling)
(Woodcock et al. 2014)	Legume-grass mixture	~ 0,09 ha	**Yes**	**Yes**	Wild Bees *Apis mellifera* Butterflies Hoverflies	Several crops	5 years (4 year sampling)
(Pywell et al. 2006)	Fabaceae (white and red)	6 m width	**Yes**	**Yes**	Bumble bees	Cereals	1 year
(Potts et al. 2009)	Legume-grass mixture	50 × 10 m	**Yes**	**Yes**	Bumble bees	Barley	4 years
(Potts et al. 2009)	Complex mixture (legumes, kale, quinoa, mixed cereals and linseed)	50 × 10 m	**Yes**	**Yes**	Butterflies	Barley	4 years

[a] The significant effect on the occurrence of individuals and species was higher in legume-forb-grass mixtures than in legume-grass mixtures

8.4.3 Feasibility and acceptability by farmers

A total area of approximately 1.38 million hectares of agricultural land were declared as EFA by German farmers in 2016 (Hemmerling et al. (2016), Table 51). Areas with catch crops and green cover (68%), land lying fallows (15%) and nitrogen-fixing crops (13%) contribute to more than 95% of the total EFA area in Germany (Table 51, Figure 33). Landscape features (i.e. trees, hedges, ponds, field margins, stone walls) accounted for 2.2% only. The total area of field edges, buffer strips, strips along forest edges, afforested areas and short-rotation coppice was < 2%.

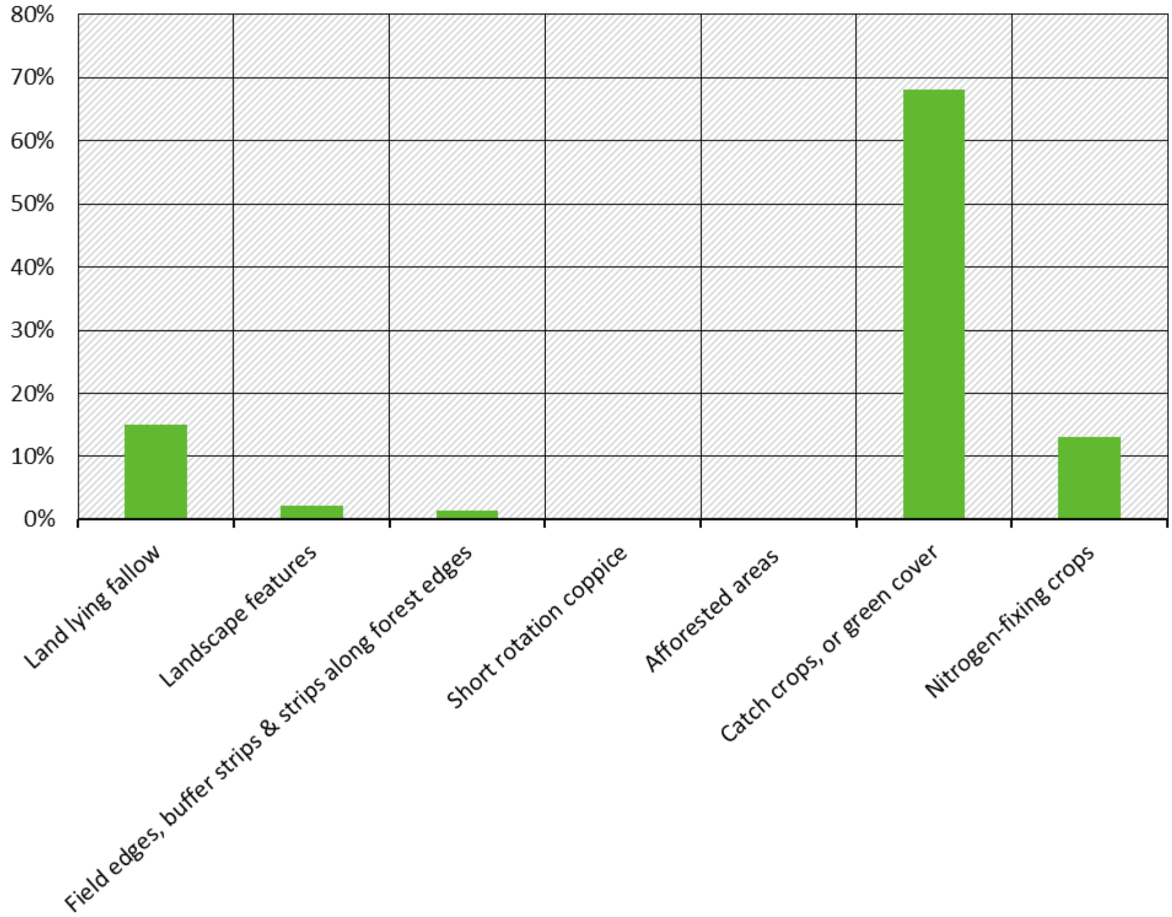
Based on a survey of 850 German farmers (with > 30 ha arable land) conducted in June 2016 on behalf of the German Farmers' Association (DBV), 83% of farmers declared ecological focus areas in the year 2016 (Hemmerling et al. (2016); Figure 34). The most frequently declared type of EFA was catch crops/green cover areas, which were declared by 56% of the farmers. Significantly fewer farmers declared EFAs of the type's landscape features (33%), land lying fallows (27%), field edges, buffer strips, strips along forest edges (20%), nitrogen fixing crops (17%). No data were reported for terraces, short rotation coppice and afforested areas.

Table 51: Ecological focus areas in Germany in hectares in the year 2016 based on data from German BMEL (areas for each EFA in ha taken from Hemmerling et al. (2016)).

Type of ecological focus area	Area (unweighted) [ha]	Proportion of total EFA (unweighted) [%]
Land lying fallow	209 265	15%
Terraces	n.r.	-
Landscape features	30 549	2.2%
Field edges, buffer strips & strips along forest edges	20 855	1.5%
Short rotation coppice	2 474	0.2%
Afforested areas	975	0.1%
Catch crops, or green cover	938 374	68%
Nitrogen-fixing crops	175 646	13%
Total area	**1 378 138**	

n.r.: not reported.

Figure 33: Proportion of total ecological focus areas in Germany (unweighted) in the year 2016 based on data from German BMEL (calculated from data taken from Hemmerling et al. (2016)).

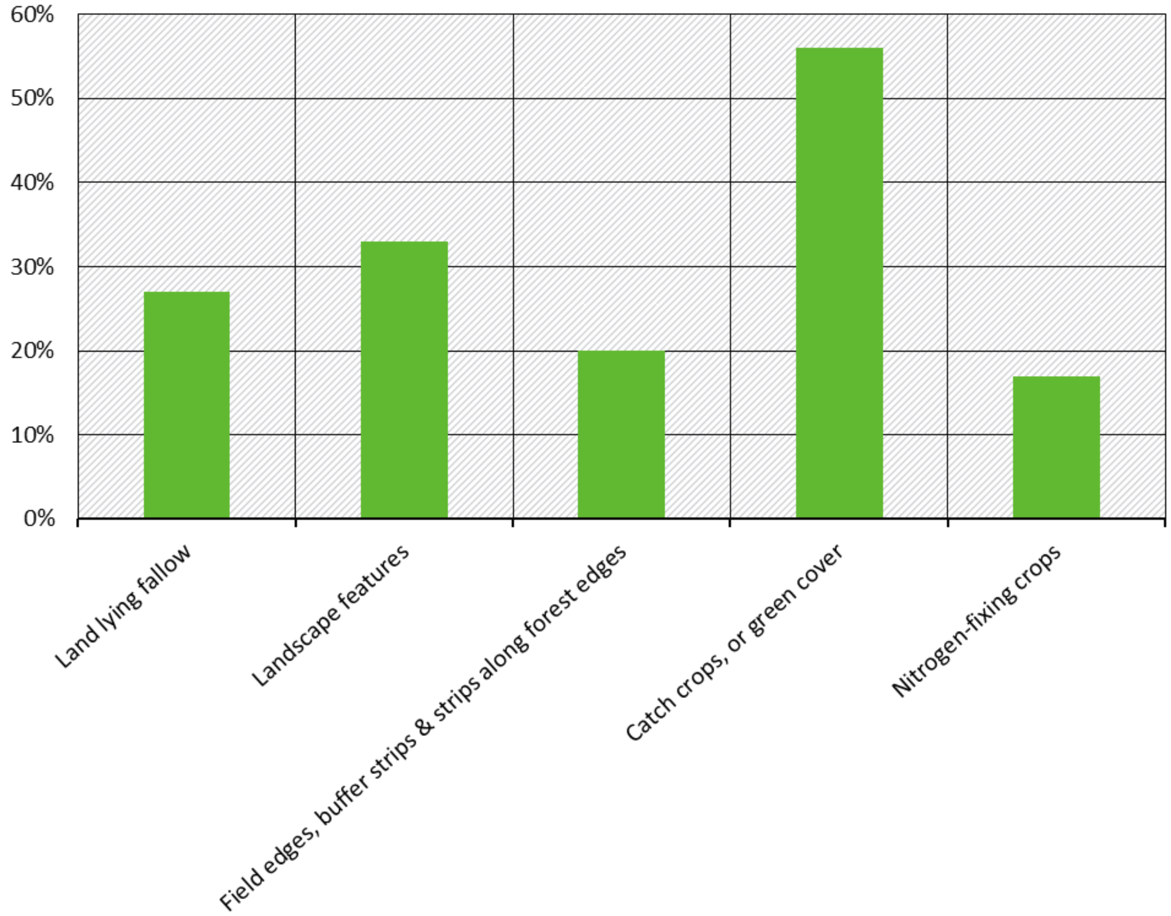


Footnote: no data reported for terraces.

Source: own illustration, ecotox consult (calculated from data taken from Hemmerling et al. 2016)

In general, the feasibility and acceptance are important factors for the declaration of an EFA by farmers. Particularly the acceptability of areas with catch crops or green cover (68% of total EFA area, declared by 56% of German farmers; Table 51, Figure 33 and Figure 34) and nitrogen-fixing crops (13% of total EFA area, declared by 17% of farmers) is considered to be high. It could be assumed that the high acceptability of these EFA types is based on the fact that these areas do not restrict the agricultural production but even positively influence the agricultural land by enhancing the yield of the main crop (Vogt-Kaute 2013). Farmers can continuously cultivate their main culture without taking land out of production and are furthermore not very restricted in agricultural management (Ehlers et al. 2014), e.g. the application of pesticides in nitrogen-fixing crops and crops undersown with grasses (green cover) is allowed all-season. In catch crops the application is only allowed in the in the following year. So, an important factor in the decision of farmer is, whether the EFA is a productive area or an area out of production, as also pointed out by Pe'er et al. (2017).

Figure 34: Frequency of German farmers that declared ecological focus areas in the year 2016. Data based on a survey of 850 farmers (with > 30 ha arable land) conducted in June 2016 on behalf of the German Farmers' Association (DBV; data from Hemmerling et al. (2016)).



Footnote: no data reported for terraces, short rotation coppice and afforested areas.

Source: own illustration, ecotox consult (data from Hemmerling et al. 2016)

With 15% of total EFA area land lying fallow was the second largest and the third most frequently declared (by 27% of farmers) EFA type in Germany in the year 2016 (Hemmerling et al. (2016), Table 51, Figure 33 and Figure 34). According to Nitsch et al. (2016) also in previous years, land lying fallows were a popular measure in Germany. As described in section 8.2.3.3 it can be assumed that the establishment of land lying fallow is easy feasible and also acceptability by farmers might be moderate to high. This is underlined by Jacot et al. (2002) who interviewed farmers about their experiences with fallows. Seventy-five percent of all interviewed farmers assessed the effort of implementation of general fallows as relatively low. In general, after the establishment of fallows no extensive conversation measure is needed (only regular monitoring and occasional mowing). However, farmers mentioned the growth of unintentional weeds (also indicated by (Sattler & Nagel 2010)) because of the official restrictions not to remove them with herbicides as a main maintenance problem. The growth of grassy plants is rising with the age of the fallow (from third year) but it was assessed as harmless by farmers. According the survey of Jacot et al. (2002) 70% to 80% of the Swiss population (farmer and non-agricultural people) supported the establishment of flower diverse fallows. Also the image of farmers would benefit by the implementation of fallows.

As mentioned in section 8.2.3.3, the acceptability of fallows by farmers might be highly dependent on the productivity of the agricultural land. At sites with low agricultural production farmers might be more poised to take land out of production (Sattler & Nagel 2010; Nitsch et al. 2016; Pe'er et al. 2017). Furthermore, at places with lower land-rental prices, acceptability to leave land uncropped would increase (Pe'er et al. 2017).

Based on a survey of 850 farmers landscape features (e.g. hedges, ponds, ditches, field margins) are declared by 33% of German farmers in the year 2016 (Hemmerling et al. (2016), Table 51, Figure 33 and Figure 34). Landscape features and terraces are protected under the European cross compliance regulation (Nitsch et al. 2016). With cross compliance regulation different EU-payments are linked to the compliance and maintenance of environmental related requirements. According to cross compliance, it is prohibited to remove existing landscape features i.e., hedges, trees, field margins, stone walls, ponds and ditches as well as terraces, because they fulfil important services for the environment. Therefore, it can be assumed that the required high effort associated with restrictions with respect to landscape features according to EU cross compliance regulation might distract farmers from the creation or recreation of new landscape features which is in agreement with Pe'er et al. (2017).However, if landscape features are already present in the farmland, farmers have very low effort to declare it as EFA. In this case acceptability by farmers and feasibility is considered to be rather high. This assumption is reflected by the high frequency German farmers declaring landscape features as EFA (33% of farmers in 2016; Figure 34). Although one out of three German farmers using this EFA type, landscape features representing only ca. 2% of the total EFA area in Germany in the year 2016 (Hemmerling et al. (2016), Table 51, Figure 33). The relatively low proportion of the total German EFA area could be explained by several factors:

Landscape features are usually small in size compared to areas with catch crops/green cover, nitrogen-fixing crops or land lying fallows.

- ► The coverage of actually available landscape features in Germany might be low. Thus, only a few features are eligible and can be declared as EFA by farmers (Hemmerling et al. 2016).
- ► Motivation of farmers to create new landscape features might be low because the implementation requires high efforts due to restrictions with respect to landscape features according to EU cross compliance regulation.
- ► Often the ownership and right to declare a landscape features is not clear and therefore farmers decide against a registration of such a type of EFA (Pe'er et al. 2017).

Buffer strips, field edges and strips along forest edges represent 1.5% of total EFA area in Germany in the year 2016 (Hemmerling et al. (2016); Table 51, Figure 33). In general, the implementation of temporary buffer strips, field edges or strips along forest edges might be easily feasible as presumed by Reichenberger et al. (2007) and Bereswill et al. (2014). However, when creating permanent strip elements the feasibility might be rather difficult because the effort is assumed to be higher and long-lasting losses of agricultural crop area has to be expected (Bereswill et al. 2014). Based on data from the survey of the German Farmers' Association (Hemmerling et al. 2016) the acceptability by farmers could be considered to be moderate: Approximately one out of five German farmers declared strips as EFAs in the years 2015 and 2016. However, the proportion of famers decreased slightly from 24% in 2015 to 20% in 2016 (Hemmerling et al. (2016), Figure 34), which might be linked to a decreasing acceptability to implement strips as EFA. When asked why they did not declare strips as EFA the most frequently reasons given by farmers were (Hemmerling et al. 2016):

- ► operational aspects such as "not needed / alternatives available (e.g. catch crops, fallows etc.)" and "no possibilities";
- ► non-operational (external) aspects such as "complicated requirements", "high effort", "high risk of sanction", "authorities advice against it" and "insufficient information".

However, for a sound evaluation of the time dependent development of acceptability a more detailed analysis over a longer period would be necessary. In addition, also Pe'er et al. (2017) report that farmers are possibly afraid of repayments or fines because of complicated requirements with respect to the management of such strip elements (e.g. strict minimum and maximum widths are required).

Afforested areas and areas planted with short rotation coppice representing only 0.1% and 0.2%, respectively, of the total EFA area in Germany in the year 2016 (Hemmerling et al. (2016), Table 51, Figure 33). Thus, a low acceptability by farmers could be assumed. Nonetheless, reasons for the low acceptance of short rotation coppice might be for example the low weighting factor of 0.3, which is resulting in a low area-related funding via the greening-premium of the 1st pillar of the EU CAP. Furthermore, farmers might be concerned about a potential yield decrease on neighbouring fields caused by shadowing effects of short coppices and afforested areas or a lack of water due to high water consumption of short rotation coppices and afforested areas (Nitsch et al. 2016). In addition, the creation of such EFAs is always related to a long period of time (trees have to grow). For example, areas with short rotation coppice are usually three to ten years present on the agricultural landscape (Nitsch et al. 2016). Hence, this arable land cannot be used for cultivation of e.g. major crops for a long time. The creation of short rotation coppices or afforested areas might require higher effort. However, we would assume that the management in the following years requires less effort than conventional agriculture. Therefore, the feasibility is assumed to be rather easy.

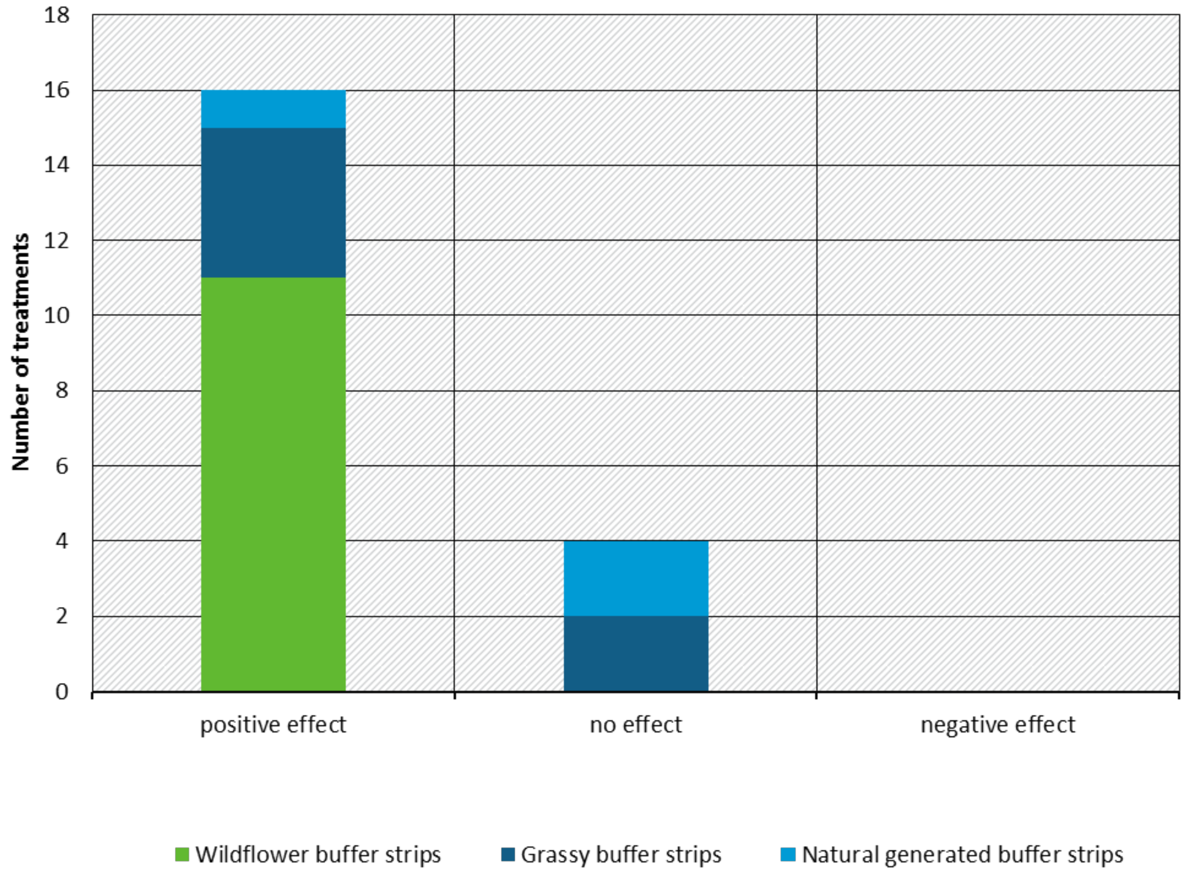
8.4.4 Minimum requirements for ecological focus areas

When implementing landscape-related mitigation measures, such as EFAs, a central question is which minimum size of such an area is necessary to compensate population-relevant effects of pesticides on FVIs in agricultural landscapes. Also other aspects, such as the number of EFAs or the life-time of biotopes might be important.

The size of an EFA is an important characteristic. In general, with increasing size of an area boundary effects are reduced (e.g. boundary effects caused by pesticide drift entries). Furthermore, greater areas provide more foraging and nesting habitats. As pointed out in section 8.2.1.2 wider strips might be more efficient in promoting FVIs abundance/diversity (Backman & Tiainen 2002; Cole et al. 2015).

However, FVIs might already benefit from small widths of buffer strips. Based on the literature reviewed in section 8.2 twenty in-field buffer strips of different widths seeded with wildflower or grasses as well as natural regenerated strips were investigated (Table 43). In total sixteen treatments showed positive effects (11 x wildflower strips, 4 x grassy strips, 1 x natural regenerated strip). Four treatments showed no effects (2 x grassy strips, 2 x natural regenerated strips) and not a single treatment demonstrated negative effects of in-field buffer strips on FVIs (Figure 35).

Figure 35: Effects of wildflower, grassy and natural regenerated in-field buffer strips on FVIs. The number of treatments demonstrating positive effects on FVI abundance and/or species richness, no effects or negative effects are shown. Data derived from Table 43. Note, that the number of treatments do not correspond to the number of studies, because some authors investigated different treatments of in-field buffers in their study.



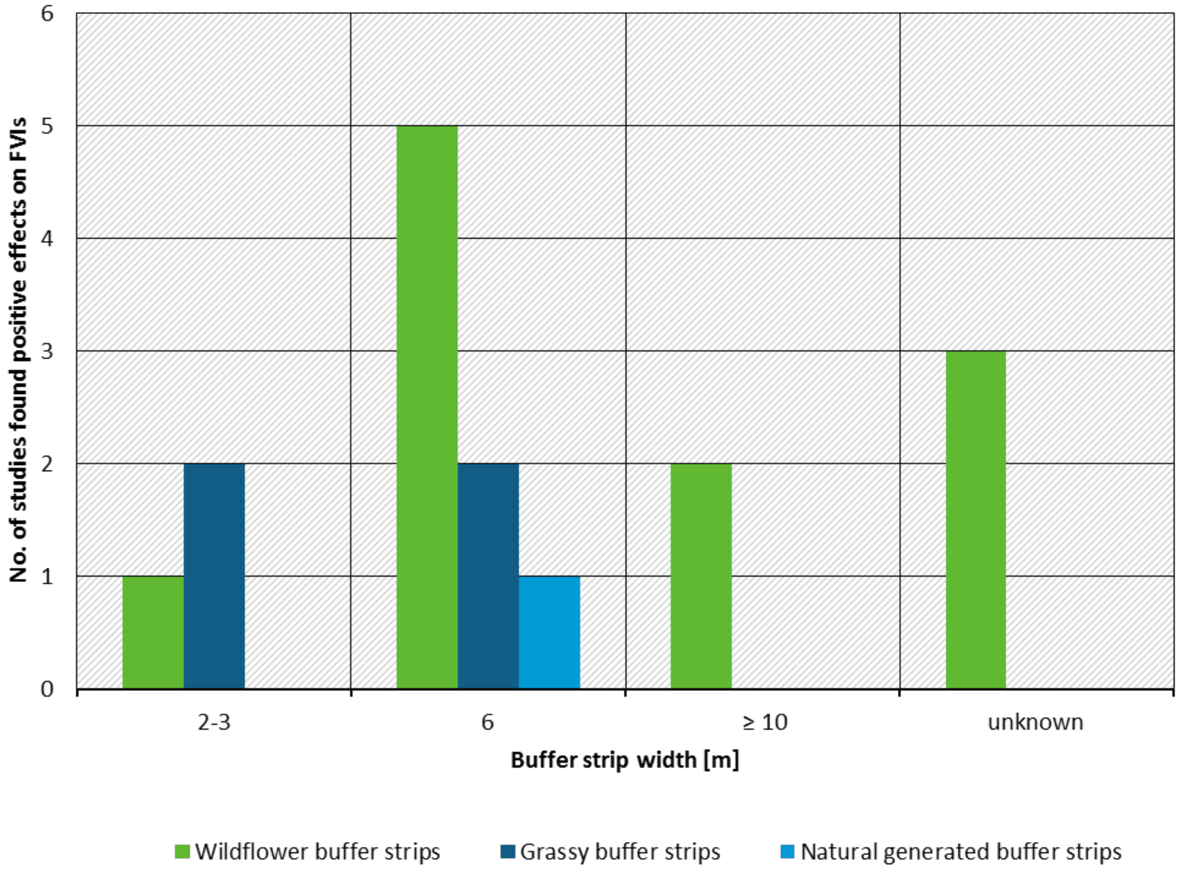
Source: own illustration, ecotox consult

Positive effects on certain FVI groups (bees, butterflies and hoverflies) could be shown three-times for buffer strips with a width of 2-3 m (Field et al. 2007; Haenke et al. 2009; Scheper et al. 2015); eight-times for 6 m wide strips (Meek et al. 2002; Carvell et al. 2004; Field et al. 2005; Pywell et al. 2005; Marshall et al. 2006; Carvell et al. 2007; Haenke et al. 2009) and, two-times for strips with a width of ≥ 10 m(Kohler et al. 2008; Haenke et al. 2009) (Figure 36).

Although FVIs might already benefit from small widths of buffer strips (e.g. 2-3 m), most available studies in literature showed positive effects of buffer strips with a width of 6 m (5 x wildflower strips, 2 x grassy strips, 1 x natural regenerated strip; Figure 36). Thus, based on available data base we assume that wildflower buffer strips adjacent to field crops with a minimum width of 6m seem to be a sufficient to support a variety of FVI species. However, a width of 6 m might be recommended also for grassy and natural regenerated buffer strips due to the demonstrated positive effects of these strips. The proposed width of 6 m is in agreement with Nitsch et al. (2016) who recommend a minimum width of 5 m for an effective promotion of FVI communities in off-field habitats.

So far, effects of buffer strips adjacent to permanent crops (fruit trees, vineyards) on FVIs are not investigated. However, pesticide drift occurring during the application process in these cultures is greater than in field crops. Therefore, adjacent EFAs might receive higher pesticide entries, and it is currently unclear if the proposed widths of 6 m are sufficient to promote FVIs in these cases. Due to this data gap, further research is needed to assess the effect of buffer strips adjacent to permanent crops on FVIs.

Figure 36: Positive effects of wildflower, grassy and natural regenerated in-field buffer strips on FVIs in relation to the buffer strip width in [m]. Data derived from Table 43.



Footnote: Flower rich agri-environment scheme investigated by Wood et al. (2015) with unknown width is categorised as wildflower buffer strip. Furthermore, wildflower buffer strip with width of 6 m includes the study by Haenke et al. (2009) who found positive effects on FVIs of 3-6 m wide flower strips.

Source: own illustration, ecotox consult

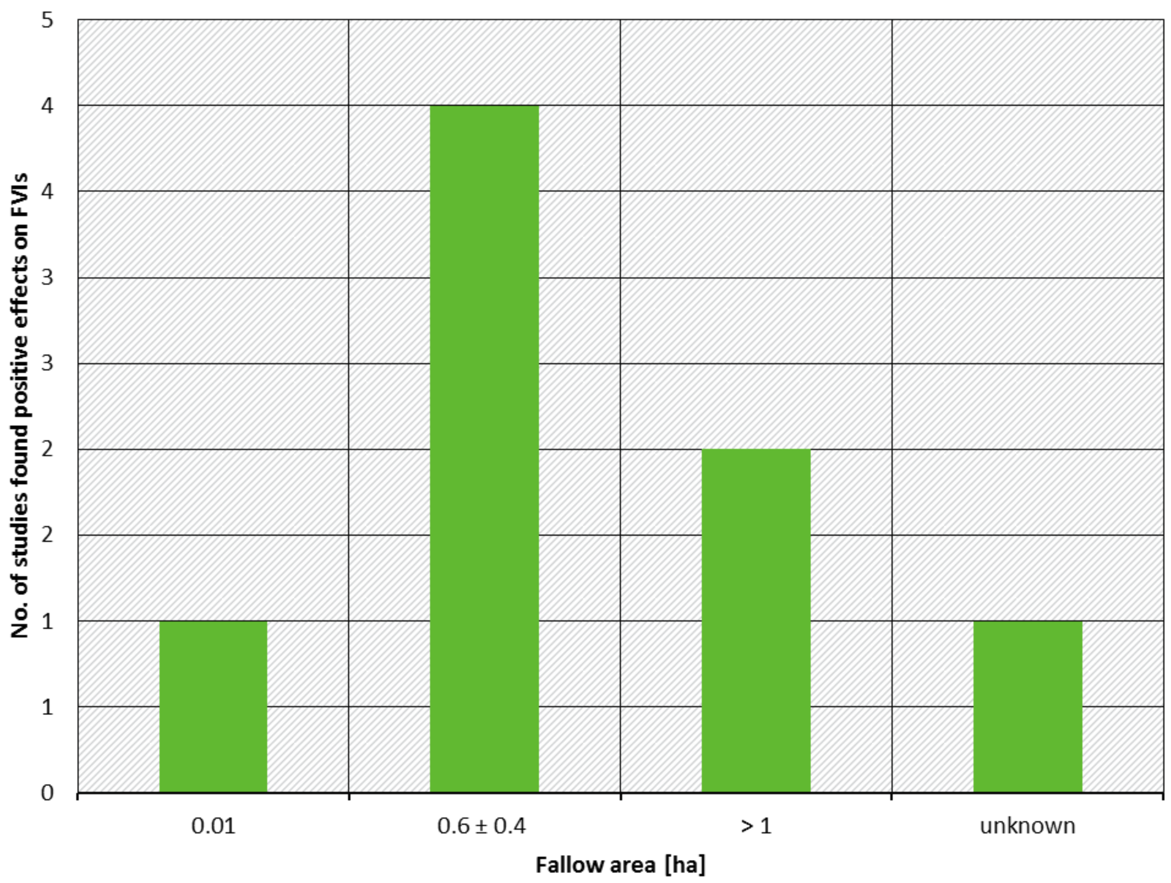
Regarding land lying fallows (uncropped arable land) nearly all investigated fallows (eight studies) had positive effects on certain FVI groups (please refer to sections 8.4.2.1 and 8.2.3). Only Gathmann et al. (1994) demonstrated no effects of fallows sown with *Phacelia* plants on FVIs.

Positive effects on certain FVI groups could be shown onetime for fallows with an area of 0.01 ha (Kohler et al. 2008), four-times for areas of 0.6 ± 0.4 ha (Gathmann et al. 1994; Steffan-Dewenter & Tscharntke 2001; Alanen et al. 2011; Kuussaari et al. 2011) and, two-times for areas > 1 ha (Denys & Tscharntke 2002; Holland et al. 2015) (Figure 37). One study is available where no information of fallow size is provided (Diekotter et al. 2006) (labelled as “unknown”, (Figure 37)). Furthermore, one study found positive effects when increasing the amount of surrounding uncropped land (Holland et al. 2015). Holland et al. (2015) showed significant increases in wild bee density and abundance or species richness of butterflies when the proportion of uncropped land is increased from 0-3% to 7,5-

10%. However, for example Gathmann et al. (1994) found no significant influences on wild bees and wasps when increasing the fallow size from 0.2 to 0.7 ha. Denys & Tscharntke (2002) found significant effects due to the fallow size on moths and hoverflies when comparing large > 1 ha fallows and small 3 m wide field margins.

Although FVI communities might already benefit from small-sized fallows (e.g. 0.01 ha) most available studies in literature showed positive effects of fallows with an area of 0.6 ± 0.4 ha (Figure 37). Thus, based on available data we assume a minimum area of 0.6 ± 0.4 ha land lying fallow seem to be a sufficient to support a variety of FVI species.

Figure 37: Positive effects of land lying fallows on FVIs in relation to the fallow sizes in [ha] (fallow types not differentiated due to the small data base). Data derived from Table 45.



Source: own illustration, ecotox consult

In general, the relation between the amount of agricultural field and the amount of uncropped areas is another important aspect for enhancing the FVI species richness and abundance in agricultural landscape. Holland et al. (2015) investigated effects on FVIs when increasing the percentage of uncropped land (0 to 10%) in the study region. Based on the results of this study, the authors recommend leaving at least 7.5% of the agricultural land uncropped. Furthermore, according Nitsch et al. (2016) a percentage of 10% total EFAs per farmer is recommended for an effective protection of biodiversity. However, the scientific data base of the recommendation by Nitsch et al. (2016) is not comprehensible. In this context Cormont et al. (2016) investigated if 3-7% natural habitat elements (defined as land out of production; i.e. linear and non-linear landscape elements, woody, herbaceous and wetland elements) within agricultural landscapes are sufficient to preserve FVI species like butterflies, hoverflies and dragonflies. The authors showed good potential to harbour many species and thus preserve farmland biodiversity with the percentage of 3% to 7% natural habitat elements.

Farmers with more than 15 ha arable land, are obliged to designate at least 5% of their arable land as EFA in order to obtain direct payments through the 1st pillar of the EU Common Agricultural Policy. The obliged minimum percentage of 5% might be too small to compensate sufficiently negative effects by the use of pesticides particularly if the vegetation is of low quality as also suggested by Kleijn & van Langevelde (2006). Kleijn & van Langevelde (2006) showed that species richness of FVIs in agricultural landscapes might depend on the quality of semi-natural habitats (i.e. forests, heather, swamps or reed beds) in the surrounding area. In this study wild bee species were positively significantly affected by high flower abundances when few semi-natural habitats were available in the landscape. The authors assumed, this might be due to the low quality of semi-natural habitats. In contrast, hoverfly species were more attracted by high abundance of flower abundance when the agricultural landscape consists of many semi-natural habitats. With respect to the recently discussion of raising the currently required amount of arable land designated as EFA from 5% to 7% in EU member states (BMEL 2015b), it should be noted that farmers in Switzerland are obliged to designate at least 7% of their arable land as EFA (Kleijn et al. 2006; Albrecht et al. 2007; Junge et al. 2009). However, in a recent report the European Commission finally recommended in March 2017 not to increase the amount of EFA to 7% (EC 2017a). The justification of this decision was that the success of EFAs to improve biodiversity would also be dependent on the quality of the respective area not only on the quantity. Moreover, the European Commission argued that farmers in the past two years anyway declared much more than 5% of agricultural land (approximately twice as much) as EFA (EC 2017a). However, based on the available data described above it is not clear, whether percentages of 5% or 7% or even higher might be sufficient for improving FVI communities; especially when farmers are mainly declaring EFAs where agricultural production is still allowed (Hemmerling et al. 2016; EC 2017a; Pe'er et al. 2017). Thus, a clear recommendation with respect to FVIs cannot be derived from the available data.

The life-time of a biotope in agricultural landscapes can be of high importance for a sufficient support of FVIs. Nitsch et al. (2016) suggest that ecological focus areas should be established for at least one vegetation period[11] . In addition, the authors assume when farmers leave the EFA over the winter or for several years on the same location, the ecological value would rise as well. However, the data base for this assumption is not comprehensible. Several authors reported positive effects on FVI communities by fallows which were established for several years (Gathmann et al. 1994; Steffan-Dewenter & Tscharntke 2001; Denys & Tscharntke 2002; Alanen et al. 2011; Kuussaari et al. 2011). In these studies, with except in the study by Denys & Tscharntke (2002), the authors showed that species richness or abundance of FVIs increased significantly with successional age. Only Denys & Tscharntke (2002) found no significant differences between 1 year fallows and 6 year fallows on moths and hoverflies. Nevertheless, particularly, Lepidoptera species seem to be significantly promoted by older fallows because they need more time for adaption to new habitats as described in Alanen et al. (2011) and Kuussaari et al. (2011). In contrast, some FVI species (e.g. bumble bees) already benefit from annual fallows. Based on the available data, FVI communities might benefit from a mixture of annual and perennial fallows present in agricultural landscapes.

In-field buffer strips (sown with wildflower/grassy mixtures or natural regenerated) were effective even after one year (one vegetation period) (Meek et al. 2002; Pywell et al. 2005; Marshall et al. 2006; Kohler et al. 2008; Haenke et al. 2009) (Table 43). However, as already outlined for fallows perennial strips have e.g. the advantage to provide habitats also in winter and there is generally more time available for species to colonize the buffer strip (Fenchel et al. 2015). This is in agreement with Haaland & Bersier (2011) who reported that the duration between sowing the seed mixture and

[11] Defined as the time where plants are growing and developing (April to September). In this time pollen and nectar supply is available.

ploughing the strip would be often too short for an adequate use as larval/egg habitat. Larval or eggs are getting destroyed in strips which are ploughed in late autumn or early spring. For this, in addition to annual in-field buffers the presence of perennial buffers is important. Perennial in-field buffer strips are funded in rural development programs of German federal states discussed in section 8.5.3.

Also the connectivity of biotopes in agricultural landscapes should be taken into account. Numerous studies found positive influences on FVI communities when a connection of EFAs was available (Sutcliffe et al. 2003; Diekotter et al. 2006; Kohler et al. 2008; Holzschuh et al. 2010; Carvell et al. 2011; Lentini et al. 2012; Balfour et al. 2015; Denisow & Wrzesien 2015; Toivonen et al. 2015; Toivonen et al. 2016). A comprehensive connectivity of diverse landscape elements (e.g. of ditches, ponds, field margins, generally flower-rich fields) in extensive managed agricultural areas is important and might promote the long-term persistence (Vasseur et al. 2013) and the movement of FVI populations within agricultural landscapes (Sutcliffe et al. 2003). Particularly Lepidoptera and wasps were shown to be negatively influenced in their mode of life by habitat isolation (Sutcliffe et al. 2003; Holzschuh et al. 2009; Holzschuh et al. 2010). In contrast social FVI species (e.g. honeybees, bumble bees) with large foraging distances were shown by Steffan-Dewenter & Tscharntke (2002) to be not much limited in their movement, even in landscapes with a low connectivity of habitats. An higher amount of organic managed fields[12] and increased crop edge densities can enhance the habitat connectivity for FVI (Holzschuh et al. 2010). Furthermore more foraging opportunities and higher interaction between FVI groups can be reached by an increased landscape heterogeneity and a higher proportion of semi-natural habitats in managed agricultural areas (Pywell et al. 2006; Rundlof et al. 2008; Holzschuh et al. 2010; Fabian et al. 2013; Potts et al. 2015; Soderman et al. 2016). For example, Mueller & Dauber (2016) found that microphagous hoverflies[13] in fields planted with *Silphium perfoliatum* (which might act as perennial food resource for FVIs in the late season) could benefit from surrounding ditches, brooks, forest edges and maize fields. In this study, species richness and abundance of microphagous hoverflies increased significantly with increasing proportion of semi-natural habitats in the surrounding area. This effect could not found for zoophagous hoverflies[14]. The implementation of a variety of different types of EFAs contributes to a more heterogeneous agricultural landscape. If these areas are evenly distributed in the agricultural landscape the biodiversity (including FVIs) might additionally promoted as assumed by Nitsch et al. (2016).

Flower-visiting insects inhabiting EFAs might also benefit from surrounding forests, nature reserve and other adjacent (semi)-natural habitats. Several authors showed that FVI species were positively influenced by surrounding forests (Fabian et al. 2013; Bailey et al. 2014; Denisow & Wrzesien 2015; Toivonen et al. 2015; Toivonen et al. 2016) or the presence of semi-natural nature reserves (Goulson et al. 2002; Kohler et al. 2008; Balfour et al. 2015). However, there are also indications that forest edges could also act as barrier which prevents the distribution of less mobile FVI species in-field (Diaz-Forero et al. 2011).

Particularly specialized or less mobile butterfly species are dependent on the surrounding semi-natural landscape since they have poor distribution properties with specific refuge needs (Toivonen et al. 2016). In this context, Denisow & Wrzesien (2015) recommend that the maximum distance of implemented field margins should not exceed a maximum distance of 1000 m to surrounding forests or meadows. The authors showed that plant species attracting bees highly benefit from nearby nature areas. Furthermore, the plant species richness decreased with increasing distance to the nature reserve which indirect influences FVI populations. Balfour et al. (2015) found that the abundance of FVIs significantly decreased when the distance to the nature reserve was rising. Kohler et al. (2008) showed a 45% decrease of hoverfly species richness and a 60% decrease of hoverfly abundance over a

[12] The federal government decided the national action plan aiming among others to reach 20 % organic managed fields globally in agricultural landscapes (BMEL 2013)

[13] Hoverflies with larvae stage feeding on dead or organic matter.

[14] Hoverflies with larvae stage feeding on aphids.

distance of 300 m to a nature reserve. These results are confirmed by Bailey et al. (2014) who showed that abundance and species richness of wild bees (i.e. *Nomada*, *Andrena* and other bees) within an oilseed rape field significantly increased with decreasing distance of the field to the forest edge (0 to 200 m). The authors presume that forest edges might be important nesting or mating habitats and should therefore be taken into account by farmers. The results of Kohler et al. (2008), Bailey et al. (2014) and Denisow & Wrzesien (2015) support the assumption that semi-natural habitats might be important elements and FVIs benefit from it when these biotopes are available in the vicinity.

In short: Minimum requirements for ecological focus areas

- It can be assumed that wildflower/grassy and natural regenerated buffer strips adjacent to field crops with a minimum width of 6 m seem to be sufficient to support a variety of FVI species. In addition to annual in-field buffers the presence of perennial buffers could be considered to be important.
- It can be assumed that a minimum area of 0.6 ± 0.4 ha land lying fallow seem to be sufficient to support a variety of FVI species. Furthermore, FVI communities might benefit from a mixture of annual and perennial fallows present in agricultural landscapes.
- Based on available date it is not clear, whether percentages of 5% or even higher might be sufficient for improving FVI communities. It can be assumed that besides the quantity, the quality of EFA (i.e. quality of food/foraging and nesting resources) plays an important role.
- The connectivity of different EFA types might be a very important factor for the long-term persistence and mobility of FVI.
- Specific FVI groups might benefit from EFAs which are surrounded by forests, nature reserves or other (semi)-natural elements.

8.4.5 Are additional legal requirements necessary for ecological focus areas?

The main aim of EFAs is the permanent protection and promotion of biodiversity in agricultural landscapes (EU 2013; BMEL 2015b) this includes amongst others also the diversity of FVIs. Applied pesticides in agricultural landscapes are suspected as one factor for the declining of biodiversity (Marshall & Moonen 2002; Brittain et al. 2010; Balmer et al. 2013; BMEL 2013). The pesticide action-network (PAN) has described the correlation between pesticide entries (in soil or surface water) and the loss of biodiversity (PAN Germany 2016). In this context PAN Germany criticized that responsible authorities missed to develop sustainable solutions for reducing negative impacts of pesticides (PAN et al. 2012; PAN 2015). Furthermore, in 2013 the German government has developed a national action plan (NAP) for the sustainable application of pesticides. The national action plan aims amongst others at reducing the risks of pesticide use to ecosystems in order to promote biological diversity. To meet this goal, several options are proposed. This includes for example the reduction of pesticide exposure to FVIs or the establishment of measures like hedges, fallows and wildflower strips in agricultural landscapes which provide habitat possibilities for beneficial organisms (e.g. FVIs) (BMEL 2013). Also the implementation of EFAs without application of pesticides are explicitly named (BMEL 2013).

The application of pesticides is prohibited in most EFAs (e.g. land lying fallows, buffer strips or landscape features). However, there are some EFA types (i.e. nitrogen-fixing crops, short rotation coppice, areas with catch crops or green cover as well as afforested areas) where the application of pesticides is not prohibited in Germany (BMEL 2015b). In catch crops and areas with short rotation coppice the use of pesticides is at least restricted and prohibited in the claim year after the harvest of previous culture. Considering the facts that EFAs aim to protect biodiversity and pesticides are suspected as a contributor for loss of biodiversity, results to the conclusion that EFAs should be protected from pesticide entries. However, at least the application of pesticides should be prohibited in these areas as proposed by the NAP (BMEL 2013). For some EFA types (i.e. catch crops, green cover crops and nitrogen-fixing crops) it was currently discussed if the application of pesticides should be prohibited on EFAs (Council of the European Union 2016; Hemmerling et al. 2016). The debate resulted in June 2017 in the decision that the pesticide use on ecological areas including trips of eligible hectares along forest edges and areas with catch crops, green cover or nitrogen-fixing crops is banned(Parliament 2017).

In several countries (e.g. Switzerland, England) measures were developed (e.g. implementation of landscape structures) with surrounded buffer strips in order to promote biodiversity. An additional non-spray buffer might reduce pesticide entries in areas/landscape structures. For example in Switzerland when declaring biodiversity priority areas (BFFs), funding of some landscape features requires a surrounding buffer strip. For example, adjacent to hedges, copses or riparian woodland a buffer strip[15] with a width of minimum 3-6 m is required. In case of hedges, this buffer strip is required on both sides of the hedge. Adjacent to dry stone walls buffers with a minimum width of 0.5 m on both sites are required. Stone heaps or backfills have even to be surrounded with a buffer strip of a width of minimum 3 m.

Agri-environmental schemes implemented in England (e.g. ELS, HLS) also require certain uncultivated buffers between landscape features and crop where spraying is not allowed. For example, in case of hedges an uncultivated buffer is required extending 2 m from the centre of the hedge. The same is required for ditches: uncultivated land extending at least 2 m from the centre of the ditch has to be present. Furthermore, a protection zone of a width of minimum 1 m between the edge of the ditch and the field has to be implemented by the land owner Moreover, in case of trees present within agricultural fields in England, the application of pesticides is prohibited under the canopy within a 2 m radius around the tree.

Thus, in Switzerland and England the above described landscape features (i.e. hedges, trees, copses, riparian woodland, dry stone walls, stone heaps, backfills and ditches) are additionally protected by buffers against negative impacts such as pesticide spray drift entries. As a consequence FVIs using these habitats as foraging and nesting resources might be protected against the potential adverse effects of pesticides.

It could be assumed that an additional protection of EFAs against pesticide spray drift entries by (no spray) buffers might be useful in EU Member States. However, the legal requirement of an additional distance between crop and EFA would probably concern farmers because this would result in an additional loss of acreage for crop cultivation. Generally two different options are available to implement an additional distance between crop and EFA. First of all, the implementation of non-spray buffers could be required. In this case, low losses of crop yield are expected but no loss of acreage for crop cultivation. Secondly, the establishment of buffer strips overgrown with wildflowers/grasses could be required. In this case, there would be losses of agricultural land and consequently a loss of crop yield. However, this might trigger concerns by farmers, at least if no compensation payments for the creation/maintenance of these buffer areas are provided. For example, in Switzerland, the implementation of statutory buffer strips around landscape feature is funded. A similar approach is also present on EU-Level in case of ponds as EFA. EU Member States can decide, if a riparian vegetated strip around the pond with a width up to 10 m should be part of the declared EFA and can be funded (EC 2014). For example this is implemented in England (Anonymous 2013a). However, the compulsory implementation of uncultivated buffer strips around EFAs might result in further concerns by farmers that these buffer strips could become a biotope with need of protection in the future. Considering all these facts, the implementation of no spray buffers around EFAs seems to be the easiest way to allow the protection of EFAs against pesticide drift entries.

[15] Buffer strips have to be overgrown with grasses, herbs or covered by litter.

The use of drift reducing nozzles or specific end nozzles is another possibility to protect EFAs adjacent to the crop against pesticide drift and overspray (please refer to 8.2.7 and 8.2.8). Provided that appropriate spraying techniques (e.g. drift reducing nozzles) are available at farms, the implementation of this mitigation measure seems to be feasible and the acceptability by farmers is assumed to be high.

As outlined in section 8.4.4, to achieve a heterogeneous landscape it is important to implement a variety of different types of EFAs. However, at time EFAs in Germany are dominated by catch crops and green cover, nitrogen-fixing areas and land lying fallows. Possible reasons might be concerns of farmers about possible sanctions in case of failings when implementing certain EFA types (Nitsch et al. 2016). Hemmerling et al. (2016) reported that farmers for example mentioned too complicated requirements of strip elements as a reason why this EFA type is not implemented. This is in agreement with Nitsch et al. (2016), who reported that farmers criticised the complicated management of EFAs, particularly fallows and strip elements. These opinions of farmers reveal that the guideline for implementing EFAs should be clearly described and requirements should not be too complicated. With this regard, guidelines for implementing EFAs should be proven and revised if required. This might help to ensure that also other types of EFAs which were shown to be effective to promote FVI populations (e.g. wildflower strips, hedges) are implemented by farmers.

In short: Additional legal requirements which might be necessary

- Because EFAs aim to protect biodiversity the application of pesticides should be prohibited in these areas. In June 2017 it was decided on EU-level that the use of pesticides should be banned in all types of ecological focus areas.
- EFAs should be protected from pesticide entries. There are different options to meet this goal:
  - No spray zones around EFAs;
  - Uncultivated buffers around EFAs (e.g. as implemented in Switzerland). To increase the acceptance by farmers funding for the creation/maintenance of these buffer areas should be provided in order to compensate for losses of agricultural land and consequently a loss of crop yield;
  - The use of drift reducing nozzles or end nozzles adjacent to EFAs.
- For successful implementation of EFAs, the guideline describing how to implement EFAs should be clearly described and requirements of different EFA types should not be too complicated.

8.5 Opportunities for funding of risk management measures proposed for the protection of flower-visiting insects (FVI)

8.5.1 Common Agricultural Policy of the European Union

In 2013 the reform of the Common Agricultural Policy (CAP) of the European Union was adopted with the objective to make the agricultural sector greener and more sustainable for the future (BMEL 2014). To achieve these goals, a two-pillar system of subventions was established: EU-funds are available to support farmers via direct payments (1st pillar) and via an environment-friendly and sustainable development of rural areas (2nd pillar).

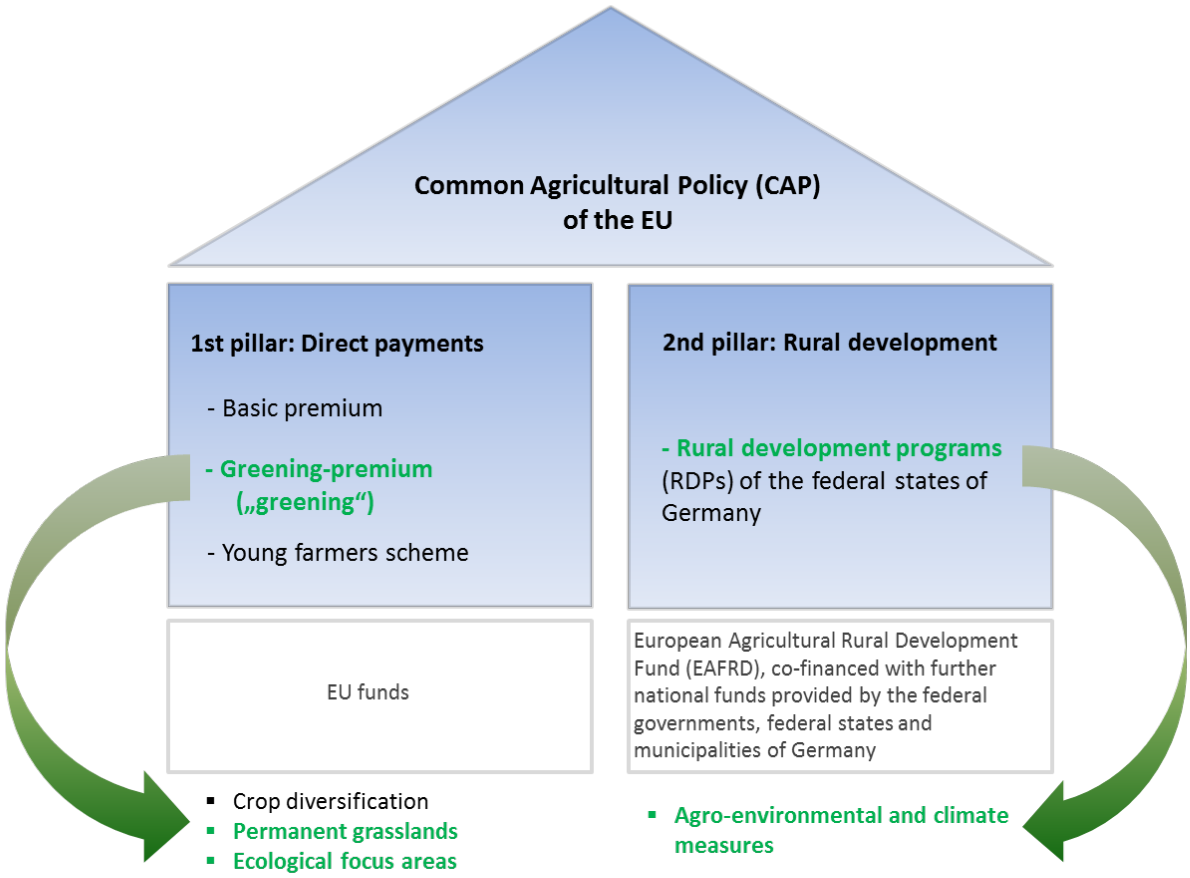
8.5.2 First pillar of the Common Agricultural Policy

The 1st pillar is composed of direct payments to farmers (Figure 38). These direct payments comprise the basic premium, the greening-premium ("greening") as well as a young farmers premium and are mandatory for all EU member states (EC 2015). Furthermore, voluntary payments, e.g. the small farmers scheme[16], can be implemented by each member state (EC 2015). All payments are subject to

[16] The small farmers scheme is implemented in Germany.

cross compliance. In order to receive payments, farmers have to fulfil certain standards with regard to environmental protection (e.g. Nitrates Directive), plant and animal health (e.g. Hormones ban Directive or the Regulation on plant protection products) and food safety (e.g. General food law) (EC 2017b). If farmers do not respect these standards payments will be reduced (EC 2017b).

Figure 38: Graphical illustration of the two pillars of the Common Agricultural policy (CAP) of the EU. Bold green: Funding opportunities of risk management measures proposed for the protection of flower-visiting insects (FVIs).



Source: own illustration, ecotox consult

For the funding opportunities with respect to the proposed risk management measures for the protection of FVI, the greening-premium plays the most important role in the 1st pillar of the CAP (Figure 38). The greening-premium has been introduced in the framework of the CAP reform to further implement environmental aspects and climate goals into the subvention policy. Farming practices contributing to these goals should be maintained and promoted. Thirty percent of direct payment budgets of the EU member states are attributed to greening and cover the three following measures (EC 2017c):

1) Crop diversification: Crop diversification measure tends to result in decreased pest pressure (i.e. reduced application of pesticides), improved farmland biodiversity and reduced soil erosion. Crop diversification applies to farmers with over 10 ha of arable land. Farmers cultivating up to 30 ha have to grow at least 2 crops and the main crop must not cover more than 75% of their arable land. In case of more than 30 ha arable land, farmers have to grow at least 3 crops, with the main crop covering at most 75% of the land and the 2 main crops covering at most 95%. Exemptions are made for farmers

who already meet the objectives of crop diversification - because a significant amount of their overall land is either grassland or fallow.

2) Permanent grasslands: Permanent grasslands are considered environmentally valuable and sensitive areas and serve as measure for preservation of off-field habitats. Environmentally sensitive permanent grasslands in Natura 2000 areas as well as outside such areas are designated by national governments. The designated grasslands cannot be ploughed or converted (e.g. into agricultural fields). Furthermore, the ratio of the permanent grassland areas to the overall agricultural area in each Member State has to be maintained and should not decline by more than 5% in comparison to the reference year (the year before). Otherwise, the respective Member State is obliged to take action to stop the decline (e.g. reversal of conversions or bans on further conversions).

3) Ecological focus areas: Farmers having arable lands of more than 15 ha must ensure that at least 5% of these areas are declared as ecological focus areas. These areas cover a broad range of biodiversity-promoting features e.g. fallow land, field margins or landscape features like hedges, trees, buffer strips as well as catch crops and nitrogen fixing crops.

Some of the risk mitigation measures proposed for the protection of FVIs in the agricultural landscape Table 42) might be funded in the context of ecological focus areas, e.g. land lying fallow or buffer strips (see also Table 52). Moreover, there are further ecological focus areas which might benefit FVIs. For more information on these ecological focus areas please refer to chapter 8.4.

Greening is obligatory to all farmers which obtain direct payments from the EU. Exceptions are made for organic farms and small-scale farms. Furthermore, if greening requirements are not fulfilled, farmers are subject to penalties e.g. reductions in payment (EC 2017c).

In Germany, the budget for the first pillar will be 4.85 billion € per year between 2014 and 2020. Greening will be granted with around 85 € per hectare and year (BMEL 2014).

8.5.3 Second pillar of the Common Agricultural Policy

The 2nd pillar comprises specific programs for sustainable and environment-friendly farming and rural development (Figure 38). The main supporting instrument in implementing the EU priorities (a high level of competitiveness in the agricultural sector, the secure sustainable management of natural resources and the support of economic strength in rural regions) for the development of rural areas is the European Agricultural Rural Development Fund (EAFRD) (BMEL 2014). Every EU member state receives an allocation of this fund by providing EAFRD support programs (hereinafter called rural development programs (RDP)). In Germany, 13 RDPs are conducted on federal state level for the funding period 2014-2020. Each federal state implement a own RDP, whereas Lower Saxony and Bremen as well as Berlin and Brandenburg established a joint program and Hamburg does not participate with a program in this period (DVS 2017). These programs mainly include voluntary environmental and climate measures related to agriculture, as well as measures to improve animal welfare and foster organic farming. These measures should contribute to a higher biodiversity, reduction of plant protection products (PPP) and fertilizer usage, improvement of soil quality, protection of soil against erosion (wind and water) and reduction of greenhouse gas emissions (BMEL 2015a).

For the actual funding period (2014-2020) the budget for these 13 programs in Germany will be around 1.35 billion € per Year co-financed with further national funds provided by the federal governments, federal states and municipalities (BMEL 2015a).

As outlined above, certain requirements related to greening, such as e.g. ecological focus areas, must be fulfilled to receive direct payments by the EU. Some agro-environmental measures are also allowed to be conducted within the ecological focus areas, besides others e.g. “integration of structural elements in the agricultural landscape” or “diversification of crops on arable land”. The federal states define which and if agro-environmental measures can be realized on ecological focus areas (BMEL

2015a). If agro-environmental measures are done on ecological focus areas subventions in the course of RDPs are reduced because the ecological focus areas are already financed by greening-payments (i.e. direct payments via the 1st pillar of the CAP). A double financing is thus avoided (BMEL 2015a).

In the following the agro-environmental measures contained in the RDPs of Rhineland-Palatinate (EULLE 2015) and Saxony-Anhalt (EPLR 2015) and the joint program of Lower Saxony and Bremen (PFEIL 2015) are exemplarily reviewed for possible funding opportunities for risk management measures (RMMs) related to the protection of flower-visiting insects (FVI) as listed in chapter 8.1. A detailed overview is given in Table 52.

The proposed agro-environmental and climate measures of the three rural development programs are mainly measures promoting biodiversity in general.

In-field buffer strips are funded in all three federal states (Table 52). In Rhineland-Palatinate grassland buffer strips for erosion control and buffer strips besides surface water bodies are funded (measure "integration of structural elements in the agricultural landscape"). Furthermore, annual and perennial flower strips ("margin- and band structures") as well as cropped strips are subsidized ("conservation management agreements for arable lands"). In Lower Saxony and Bremen annual and perennial flower strips, conservation strips, grassland buffer strips besides surface water bodies and grassland buffer strips for erosion control are subsidized whereas in Saxony-Anhalt annual and perennial flower strips and conservation strips ("integration of structure elements in fields") are promoted. All measures include specifications concerning seed mixes, plant species, stripe width as well as cultivation and management of the elements definite by each federal state (for further information see the respective RDP). The application of pesticides and nitrogen fertilizers is prohibited on all respective strips. Exceptions are made in Rhineland-Palatinate for cropped strips ("conservation management agreements for arable lands"). Here, the application of pesticides and fertilizer is allowed but should be avoided as far as possible.

High vegetation (such as hedges, shrubberies, trees) is funded in Lower-Saxony and Bremen (Table 52). In the RDP of Lower Saxony and Bremen strips for control of wind erosion and hedges for protection of birds with deciduous trees and shrubs are promoted. The application of nitrogen fertilizers and pesticides is prohibited. In Rhineland-Palatinate and Saxony-Anhalt no funding possibilities are given.

All three RDPs subsidize organic farming, which mainly means no usage of synthetic pesticides. For the reduction of the amount of pesticides applied in agricultural landscapes Rhineland-Palatinate implemented different measures: "transformation of arable land to grasslands", "alternative plant protection methods", and "biotechnical measures in plant protection in vineyards". In the course of "transformation of arable land to grassland" the application of pesticides is not permitted. Exceptions are made in case certain pest plant or rodent species occur. Within the scope of the measure "alternative plant protection methods" synthetic insecticides are supposed to be replaced by biological (e.g. *Trichogramma* against *Ostrinia nubilalis*) or biotechnical plant protection methods (e.g. glue products for control of caterpillar). The measure "biotechnical measures in plant protection in vineyards" requires the replacement of synthetic insecticides by using insect pheromones. A further possible risk management measure for reduction of the amount of pesticides is crop diversification and implemented in all three RDPs. In this measure ("diversification of crops on arable land"), a minimum of 5 different crops should be cultivated on arable lands. To obtain positive environmental effects, especially nitrogen-fixing crops or mixtures of e.g. grass with legumes should be included in the crop rotation. Based on the crop diversification with nitrogen-fixing crops a decrease in pest pressure could occur and lead to a reduction in application of pesticides. Furthermore, the use of nitrogen fertilizer is reduced and due to different crops at the same time farmland biodiversity is improved (EPLR 2015; EULLE 2015; PFEIL 2015).

No rural development program includes specific measures relating to the usage of spray drift reducing techniques.

Measures for the preservation and management of off-field habitats are mainly related to grassland. The RDPs of Rhineland-Palatinate, Saxony-Anhalt and Lower Saxony and Bremen comprise measures concerning the preservation and management of grasslands with the aim to protect and enhance biodiversity. These measures include no or restricted use of pesticides and fertilizers, regularly mowing and pasturing as well as maintenance work. No measures concerning sowing of seed mixes, maintenance of hedges or the creation of nesting possibilities for bees (e.g. bee banks) are given in the three RDPs.

Measures for the preservation of fruit orchards are promoted by Rhineland-Palatinate ("conservation management agreements for Streuobst") and Saxony-Anhalt ("funding of extensively used fruit orchards"). In Rhineland-Palatinate the preservation of traditional fruit orchards is subsidized to protect these specific species-rich habitats; e.g. preserving the trees and deadwood for breeding birds and the flowering meadows for pollinating insects. Furthermore, the maintenance of old trees (e.g. pruning of trees) as well as the planting of new trees is subsidized. The removal of trees and the application of nitrogen fertilizers and PPP are prohibited. In Saxony-Anhalt the preservation of extensively used fruit orchards is funded. Here, the removal of trees is not allowed during a commitment period.

Table 52: Detailed overview of funding possibilities of risk management measures proposed for the protection of flower-visiting insects (FVI). Comparison of the EU funding (1st pillar of the CAP: greening) with the agro-environmental measures of the rural development programs (2nd pillar of the CAP: RDPs), exemplarily shown for Rhineland-Palatinate, Saxony-Anhalt and Lower Saxony and Bremen.

Proposed risk management measures for FVI	Possibility for funding based on:			
	1st pillar of the CAP	2nd pillar of the CAP (RDP)		
	European Union	Rhineland-Palatinate	Saxony-Anhalt	Lower Saxony and Bremen
In-field buffer strips (cropped, uncropped, flower strips)	x[a]	x	x	x
Extension of small field margins	-	-	-	-
Creation of conservation fallows	x[a]	-	-	-
High vegetation (hedges, tree rows)	x[a]	-	-	x
Reduction of the amount of pesticides applied in agricultural landscapes	x[f]	x[c,d,f]	x[d,f]	x[d,f]
No-spray zones	-	-	-	-
Spray drift reducing techniques	-	-	-	-
No overspraying of off field habitats	-	-	-	-
Timing of application:				
► **application in the evening after honeybee flight**	-	-	-	-

Proposed risk management measures for FVI	Possibility for funding based on:			
	1st pillar of the CAP	2nd pillar of the CAP (RDP)		
	European Union	Rhineland-Palatinate	Saxony-Anhalt	Lower Saxony and Bremen
► no application when crops flowers or flowering vegetation between fruit tree/vineyard rows are present	-	-	-	-
► application at low wind speeds	-	-	-	-
Preservation and management of off-field habitats:	x[b]	x[e]	x[e]	x[e]
► Mowing rhythm	-	-	-	-
► Sowing of seed mixes	-	-	-	-
► Maintenance of hedges	-	-	-	-
► Creation of nesting possibilities for bees e.g. bee banks	-	-	-	-
Leaving deadwood in fruit orchards	-	x	x	-
Sowing of seed mixes between vine and fruit tree rows	-	-	-	-

x = possible management measures for the protection of flower-visiting insects (FVI) are implemented in the EU funding (greening) and/or rural development program (RDP) of the respective federal state;

- = no management measure for the protection of flower-visiting insects (FVI) is included in the EU funding (greening) and/or rural development program (RDP) of the respective federal state;

[a] = ecological focus area (landscape feature);

[b] = permanent grassland (greening measure);

[c] = alternative and biotechnical plant protection methods;

[d] = organic farming;

[e] = measures concerning the preservation and management of grasslands (e.g. regularly mowing, no or restricted use of pesticides and fertilizers, regularly pasturing, maintenance work);

[f] = crop diversification

9 Open questions & further research

Throughout this report knowledge gaps have been identified that need to be closed in order to comprehend the ecology of FVIs and estimate the effects of pesticide exposure in their habitats. This is necessary to assess the risk pesticides pose to this group and propose adequate risk mitigation measures.

It has been established that the taxonomic groups of bees, flies, moths, butterflies and beetles are relevant FVIs. However, the database for non-bee Hymenoptera and Hemiptera does not allow to assess their FVI status. Therefore, these taxa need to be studied in more detail regarding their relationships with plant communities and FVI-plant networks. This also applies to the less well-researched confirmed FVI groups moths, butterflies and beetles. To understand the interplay of plants and FVIs there is a need to collect and analyse ecological trait data (e.g. flower preferences, host plants, function of visitation), particularly for groups other than bees.

To understand how FVIs react to stress caused by pesticides in the long term, there is a need to identify the vulnerable groups among them. Firstly, it should be investigated which FVI live in habitats that overlap with agricultural areas. Secondly, ecological trait data need to be collected for all FVIs other than bees to discover relevant properties and allocate ecologically vulnerable groups. Furthermore, increasing population monitoring efforts should be undertaken to confirm the established criteria and continuously assess the status of these groups.

Potential pesticide effects are dependent of the probability and quantity of exposure. Since exposure data were almost exclusively measured in bee studies or with consideration of bee ecology, exposure of the remaining FVI groups should be researched. The ecological differences of the groups (e.g. herbivore/florivore larval stages, aboveground/underground nesting) should be taken into account. Moreover, FVIs may be exposed through several environmental matrices. However, clear quantitative links remain to be established for less well-researched matrices such as soil, plant stem/leaves, guttation water, honeydew, extrafloral nectaries or puddles. Furthermore, a comprehensive residue database in all relevant matrices (e.g. pollen/nectar, stem/leaves, soil, water) of crop and non-crop areas should be compiled. Dust dispersion after granule applications should be investigated as well as off-field pesticide loads in soil and plant matrices for a wide range of pesticides. To incorporate the inherent mobility of FVIs a landscape-scale exposure assessment needs to be developed to realistically evaluate FVI exposure by food or contact following spray and solid applications in in-field and off-field scenarios.

The assessment of pesticide effects on FVIs should rely on a broad database. Therefore, other species (from several FVI groups) than the honey bee and bumble bees need to be tested, especially regarding sublethal and field effects. There is a need to develop adequate population models for different FVI groups to define tolerable effects and an appropriate protection goal for FVI risk assessment. Direct herbicide effects on plants that may indirectly affect FVIs by reduced food plant quality, food plant presence or inflorescence have been neglected so far. This needs to be addressed with great urgency. Furthermore, pesticide impact on FVI populations and biodiversity as well as their ecosystem services remain understudied. Exerted effort should be put in the study of these more complex, broad-scale effects. Additionally, source-sink dynamics of mobile FVIs need to be incorporated in field study designs. Such landscape-scale effects should be assessed with appropriate modelling approaches that need to be developed.

Considering the efficiency of risk mitigation measures to promote FVIs in agricultural landscapes, best investigated mitigation measures are wildflower in-field buffer strips and fallows. For both the efficiency to promote FVIs was scientifically demonstrated in available literature. In addition efficiency could also be demonstrated for further risk mitigation measures (e.g. hedges, sowing of seed mixtures). However, there are many measures with further need of research. Particularly in case of application related measures (e.g. reduction of application rate, spray drift reducing techniques) there are no studies available investigating the effects on FVIs. In general, focus of available studies is the potential to reduce pesticide entries in off-field habitats or pesticide inputs in-crop. Based on the efficiency of these measures to reduce pesticide exposure, it can be assumed that also effects on FVIs are reduced. But there are also some landscape-related measures (e.g. creation of nesting possibilities, leaving deadwood in fruit orchards) where the available database in literature is insufficient. Efficiency to promote FVIs can be assumed based on ecological considerations. However, further research is necessary to scientifically confirm this assumption.

Further research need was also identified for some types of EFAs (i.e., terraces, trees, ponds and ditches, short rotation coppices, afforested areas, catch crops/greencover, nitrogen-fixing crops), because the data base documenting possible effects on FVIs of these EFA types are pretty rare.

10 List of Annexes

- Annex I: Studies documenting Lepidoptera visiting crop flowers or occurring in agricultural sites.

11 References

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11.2 References (section 7 & 8)

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Annex I

Studies documenting Lepidoptera visiting crop flowers or occurring in agricultural sites.

A) Orchards

Lepidoptera visiting crop flowers

Butterflies	Fruit set increased significantly with the abundance of butterflies (study sites near Chernobyl)	Moller et al. (2012)
Lepidoptera	1% of the flower visits on pear blossoms were Lepidoptera	Lee et al. (2007)
Lepidoptera	7% of the flower visits on apple blossoms were Lepidoptera	Lee et al. (2008)

Lepidoptera occurring in orchards

Lepidoptera/butterflies		Garcia & Minarro (2014)
Butterflies	14 butterfly species occurred in an old orchard (including surrounding habitats)	Voigt (2010)
Synanthedon myopaeformis (Borkhausen, 1789)		Aurelian et al. (2015)
Noctuidae		Aurelian et al. (2015)
Several moth families		Aurelian et al. (2015)
Lepidoptera caterpillars	found in the grass cover	Simon et al. (2007)
Papilio machaon (Linneaus, 1758)		Settele et al. (2000)
Thecla betulae (Linneaus, 1758)		Settele et al. (2000)
Satyrium pruni (Linneaus, 1758)		Settele et al. (2000)
Vanessa atalanta (Linneaus, 1758)		Settele et al. (2000)
Melitaea parthenoides (Keferstein, 1851)		Settele et al. (2000)
Lasiommata megera (Linneaus, 1767)		Settele et al. (2000)

B) Vineyards

Lepidoptera occurring in vineyards

Maniola jurtina (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Satyrium esculi (Hubner, 1804)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Polyommatus icarus (Rottemburg, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pieris rapae (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Aricia agestis (Denis & Schiffermuller, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Brintesia circe (Fabricius, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pyronia cecilia (Vallantin, 1894)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Melanargia galathea (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Melitaea didyma (Esper, 1779)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Coenonympha pamphilus (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Lycaena phlaeas (Linnaeus, 1761)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pieris brassicae (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pontia daplidice (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Lasiommata megera (Linnaeus, 1767)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Polyommatus escheri (Hubner, 1823)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Limenitis reducta (Staudinger, 1901)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Carcharodus alceae (Esper, 1780)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pieris napi (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Argynnis paphia (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Pieris mannii (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Iphiclides podalirius (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Cacyreus marshalli (Butler, 1898)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Melitaea cinxia (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011

B) Vineyards		
Polyommatus thersites (Hubner, 1834)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Mellicta athalia (Rottemburg, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Melitaea phoebe (Denis & Schiffermuller, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Brenthis daphne (Denis & Schiffermuller, 1775)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Gonepteryx rhamni (Linnaeus, 1758)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Hipparchia statilinus (Hufnagel, 1766)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Colias crocea (Geoffroy, 1785)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Melanargia occitanica (Esper, 1793)	31 butterfly species recorded in vineyards in France	Lizee et al. 2011
Butterflies and burnet moths	1-14 species in conventional vineyards; 10-38 species in integrated vineyards: 12-33 species in biological vineypards	Hluchý et al. (2009)
Iphiclides podalirius (Linneaus, 1758)		Settele et al. (2000)
Satyrium acacia (Fabricius, 1787)		Settele et al. (2000)
Scolitantides orion (Pallas, 1771)		Settele et al. (2000)
Plebeius argyrognomon (Bergsträsser, [1779])		Settele et al. (2000)
Polyommatus (Aricia) eumedon (Esper, [1780])		Settele et al. (2000)
Melitaea phoebe ([Denis & Schiffermüller], 1775)		Settele et al. (2000)
Lasiommata megera (Linnaeus, 1767)		Settele et al. (2000)
Lasiommata maera (Linnaeus, 1758)		Settele et al. (2000)
Acronicta rumicis (Linnaeus, 1758)		Hahn et al. (2016)
Agrotis exclamationis (Linnaeus, 1758)		Hahn et al. (2016)
Agrotis puta (Hübner, [1803])		Hahn et al. (2016)
Agrotis segetum ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)
Amphipyra tragopoginis (Clerck, 1759)		Hahn et al. (2016)
Apamea sordens (Hufnagel, 1766)		Hahn et al. (2016)
Autographa gamma (Linnaeus, 1758)		Hahn et al. (2016)
Axylia putris (Linnaeus, 1761)		Hahn et al. (2016)

B) Vineyards		
Deltote bankiana (Fabricius, 1775)		Hahn et al. (2016)
Discestra trifolii (Hufnagel, 1766)		Hahn et al. (2016)
Emmelia trabealis (Scopoli, 1763)		Hahn et al. (2016)
Euxoa tritici (Linnaeus, 1761)		Hahn et al. (2016)
Luperina testacea ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)
Mamestra brassicae (Linneaus, 1758)		Hahn et al. (2016)
Mythimna albipuncta ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)
Mythimna ferrago (Fabricius, 1787)		Hahn et al. (2016)
Mythimna pallens (Linnaeus, 1758)		Hahn et al. (2016)
Noctua comes Hübner, [1813]		Hahn et al. (2016)
Noctua pronuba (Linneaus, 1758)		Hahn et al. (2016)
Ochropleura plecta (Linnaeus, 1761)		Hahn et al. (2016)
Pseudeustrotia candidula ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)
Tyta luctuosa ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)
Xestia c-nigrum (Linnaeus, 1758)		Hahn et al. (2016)
Xestia xanthographa ([Denis & Schiffermüller], 1775)		Hahn et al. (2016)

C) Arable fields

Lepidoptera observed in arable fields

Colias erate (Esper, [1803])	alfalfa	Settele et al. (2000)
Colias crocea (Fourcroy, 1785)	alfalfa	Settele et al. (2000)
Pieris brassicae (Linnaeus, 1758)	cabbage	Settele et al. (2000)
Pieris rapae (Linnaeus, 1758)	cabbage	Settele et al. (2000)
Cupido argiades (Pallas, 1771)	alfalfa	Settele et al. (2000)
Acronicta rumicis (Linnaeus, 1758)	vegetable	Hahn et al. (2016)
Agrotis exclamationis (Linnaeus, 1758)	cereal	Hahn et al. (2016)
Agrotis segetum ([Denis & Schiffermüller], 1775)	cereal, vegetable	Hahn et al. (2016)
Amphipyra tragopoginis (Clerck, 1759)	vegetable	Hahn et al. (2016)
Apamea anceps ([Denis & Schiffermüller], 1775)	vegetable	Hahn et al. (2016)
Apamea monoglypha (Hufnagel, 1766)	cereal	Hahn et al. (2016)
Autographa gamma (Linnaeus, 1758)	cereal, vegetable	Hahn et al. (2016)
Cryphia raptricula ([Denis & Schiffermüller], 1775)	vegetable	Hahn et al. (2016)
Deltote bankiana (Fabricius, 1775)	cereal	Hahn et al. (2016)
Discestra trifolii (Hufnagel, 1766)	cereal, vegetable	Hahn et al. (2016)
Emmelia trabealis (Scopoli, 1763)	vegetable	Hahn et al. (2016)
Hoplodrina octogenaria (Goeze, 1781)	cereal	Hahn et al. (2016)
Luperina testacea ([Denis & Schiffermüller], 1775)	cereal, vegetable	Hahn et al. (2016)
Macdunnoughia confusa (Stephens, 1850)	cereal	Hahn et al. (2016)
Mamestra brassicae (Linneaus, 1758)	vegetable	Hahn et al. (2016)
Melanchra persicariae (Linnaeus, 1761)	vegetable	Hahn et al. (2016)
Mythimna pallens (Linnaeus, 1758)	cereal, vegetable	Hahn et al. (2016)
Ochropleura plecta (Linnaeus, 1761)	vegetable	Hahn et al. (2016)
Plusia festucae (Linnaeus, 1758)	cereal, vegetable	Hahn et al. (2016)
Pseudeustrotia candidula ([Denis & Schiffermüller], 1775)	cereal, vegetable	Hahn et al. (2016)
Xestia c-nigrum (Linnaeus, 1758)	cereal, vegetable	Hahn et al. (2016)
Lepidoptera visiting crop flowers		
Pieris rapae (Linnaeus, 1758)		Ômura et al. (1999)

C) Arable fields		
Pieris rapae (Linnaeus, 1758)		Richards et al. (2009)
Pieris rapae (Linnaeus, 1758)		Wu et al. (2016)
Lepidoptera	sampled with yellow pan traps; less than 1% of the sampled insects	Kristen (2008)
Vanessa carduii		Stanley (2013)
Butterflies	observation of 3 butterflies visiting rape blossoms	Karise et al. (2004)
Lepidoptera observed in oilseed rape fields		
Butterflies	8 butterfly species identified during transect walks	Stanley & Stout (2013)
Pieris rapae (Linneaus, 1758)		Settele et al. (2000)