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Review

Pesticide impacts on insect pollinators: Current knowledge and future research challenges

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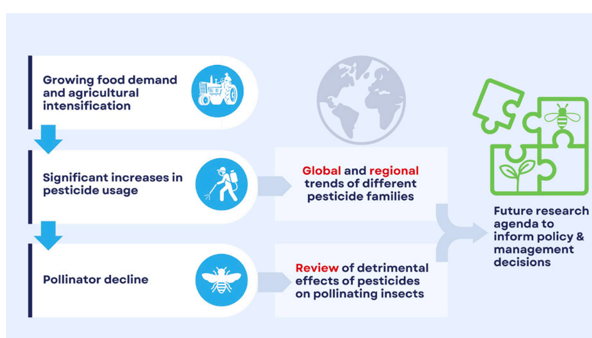
HIGHLIGHTS

- Long-term global analysis of inter-regional trends of pesticide use presented
- Updated analysis of pesticides' impact on pollinators from different global regions
- Research challenges for the future pesticide policy towards pollinator conservation outlined

GRAPHICAL ABSTRACT



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ABSTRACT

With the need to intensify agriculture to meet growing food demand, there has been significant rise in pesticide use to protect crops, but at different rates in different world regions. In 2016, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) global assessment on pollinators, pollination and food production identified pesticides as one of the major drivers of pollinator decline. This assessment

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Insecticides
Herbicides
Fungicides
Risk assessment
Policies

highlighted that studies on the effects of pesticides on pollinating insects have been limited to only a few species, primarily from developed countries. Given the worldwide variation in the scale of intensive agricultural practices, pesticide application intensities are likely to vary regionally and consequently the associated risks for insect pollinators. We provide the first long-term, global analysis of inter-regional trends in the use of different classes of pesticide between 1995 and 2020 (FAOSTAT) and a review of literature since the IPBES pollination assessment (2016). All three pesticide classes use rates varied greatly with some countries seeing increased use by 3000 to 4000 % between 1995 and 2020, while for most countries, growth roughly doubled. We present forecast models to predict regional trends of different pesticides up to 2030. Use of all three pesticide classes is to increase in Africa and South America. Herbicide use is to increase in North America and Central Asia. Fungicide use is to increase across all Asian regions. In each of the respective regions, we also examined the number of studies since 2016 in relation to pesticide use trends over the past twenty-five years. Additionally, we present a comprehensive update on the status of knowledge on pesticide impacts on different pollinating insects from literature published during 2016–2022. Finally, we outline several research challenges and knowledge gaps with respect to pesticides and highlight some regional and international conservation efforts and initiatives that address pesticide reduction and/or elimination.

1. Introduction

The area of land under agriculture has expanded over recent decades in response to growing food demands (Ramankutty et al., 2018). Continuation of this trend is projected to lead to an additional land demand of 1510 Mha by 2030 (Lambin et al., 2013; Lambin and Meyfroidt, 2011). In parallel, the per capita availability of agricultural land has been shrinking (FAO, 2020[2]), mostly in the developing world. This demand for additional land and pressure to maintain agricultural output have driven the need to intensify agricultural production, with some regions undergoing more intensive agriculture relative to others (Zabel et al., 2019). Various environmental and economic shocks to food systems due to recent and ongoing geo-political turmoil (e.g., the Russian-Ukrainian war impacts on food, fuel, and fertilizer) and effects of climate change (e.g., droughts worldwide) have further added to this pressure. With increasing intensification, there has been concomitant growth in the use of agrochemicals particularly pesticides[3] (i.e., insecticides, herbicides, and fungicides), although there is considerable variation in pesticide use across regions (Pellegrini and Fernández, 2018).

Worldwide, pesticides (along with other facets of land-use change and land-management intensification) are well established as being among the principal anthropogenic pressures linked to loss of insect populations (Osterman et al., 2019; Sánchez-Bayo and Wyckhuys, 2019; Straw et al., 2021, 2022; Straw and Brown, 2021a, 2021b; Dicks et al., 2021; Linguadoca et al., 2022; Tamburini et al., 2021a; although not all studies found negative effects, e.g., Tamburini et al., 2021b; Schwarz et al., 2022). Pesticide impacts on pollinating insects in particular have been reported consistently over recent decades both through empirical studies, meta-data analyses and syntheses (Cecala and Wilson Rankin, 2021; Douglas et al., 2022; Gill and Raine, 2014; IPBES, 2016; Kenna et al., 2019; Main et al., 2020; Raine and Rundlöf, 2024; Rundlöf et al., 2015; Stanley et al., 2016). Despite this scientific focus, various gaps in our understanding of pesticides' lethal and sub-lethal impacts on insect pollinators remain and were identified by the IPBES assessment on Pollinators, Pollination and Food Production (IPBES, 2016, ibid. p.64). Considering the crucial role of pollinating insects in plant reproduction and the fact that approximately 85 % of the main crop types around the world depend on insect pollination to varying degrees (Klein et al., 2007; Aizen et al., 2019), bridging these knowledge gaps is crucial to inform a more comprehensive pesticide risk assessment framework for safeguarding pollinating insects across the globe.

One major knowledge gap is the extent of regional variation of pesticide use and availability of standardized and/or accurate data for pesticides in different parts of the world (Dicks et al., 2021). To date research on pesticides and their impacts has been predominantly done in Europe and North America, with relatively little information from other world regions where conventional intensification of agriculture has gained momentum over the past half a century (Kuyper and Struik, 2014; Pretty et al., 2006). Given the inter-regional variation in levels of agricultural intensification and agrochemical use, the intensity and magnitude of pesticide application and associated risks are also likely to vary between regions. A recent expert-elicitation analysis in which experts scored the severity and magnitude of pressures on pollinators (Dicks et al., 2021) concluded that pesticides were among the most important causes of pollinator decline globally, but with variation in that pressure across regions. This risk to pollinators is likely growing in many world regions. This is because the latest generation of pesticides have been reported to have a greater Total Applied Toxicity (TAT).[4] A recent meta-analysis of 381 pesticides used over the last twenty-five years that considered 1591 substances with specific acute toxicity threshold values for eight non-target species groups showed a marked increase in total applied toxicity (TAT) of insecticides since 2005 (Schulz et al., 2021). This study concluded that these increased toxicities are being driven by use of certain insecticides, such as highly toxic pyrethroids and neonicotinoids (Schulz et al., 2021).

In addition to the aforementioned regional variation, there is also a major bias in the selection of non-target species for pesticide-impact studies. The greatest focus has been on the western honey bee (*Apis mellifera*) (Dirilgen et al., 2023) as a model species, and to a lesser extent, on a few species of, often managed, bumble bees (*Bombus* spp. e.g., Tamburini et al., 2021a; Wintermantel et al., 2022) and stingless bees (Bernardes et al., 2018b; da Costa Domingues et al., 2020; De Oliveira Ferreira et al., 2020; Farder-Gomes et al., 2021a, 2021b). Relatively little is known about pesticide impacts on most wild, unmanaged bee species and of those that exist, these mainly focus on the Megachilid bee *Osmia* spp. (Al Naggar and Paxton, 2021; Ii and Rangel, 2018; Mallinger

[2] Retrieved January 2023 from https://www.fao.org/sustainability/news/detail/en/c/1274219/

[3] FAO defines a pesticide as "Pesticide means any substance or mixture of substances intended for preventing, destroying or controlling any pest, including vectors of human or animal disease, unwanted species of plants or animals causing harm during or otherwise interfering with the production, processing, storage, transport or marketing of food, agricultural commodities, wood and wood products or animal feedstuffs or substances which may be administered to animals for the control of insects, arachnids or other pests in or on their bodies. The term includes substances intended for use as a plant growth regulator, defoliant, desiccant or agent for thinning fruit or preventing the premature fall of fruit, and substances applied to crops either before or after harvest to protect the commodity from deterioration during storage and transport." (FAO, 2006 - International Code of Conduct on the Distribution and Use of Pesticides. https://www.fao.org/3/bt565e/bt565e.pdf)

[4] Total applied toxicity (TAT) per species group, substance is calculated by multiplying the annually applied amount (i.e., mass) of individual pesticides with the reciprocal of the pesticide and species group–specific Regulatory Threshold Limits (RTL) per substance/, species group/, and year.

et al., 2015; Milone et al., 2021; Park et al., 2015; Rundlöf et al., 2015; Schwarz et al., 2022; Williams et al., 2015; Woodcock et al., 2017; Zhu et al., 2014) and *Megachile rotundata* (Ansell et al., 2021; Piccolomini et al., 2018; Pitts-Singer and Barbour, 2017). In addition, still fewer have studied pesticide impacts on non-bee insect pollinators (Pisa et al., 2015; Serrão et al., 2022; Uhl and Brühl, 2019).

Relatively little research has been dedicated to assessing impacts of different classes of pesticides (but see Tosi and Nieh, 2019; Schwarz et al., 2022; Tamburini et al., 2021a; Tamburini et al., 2021b; Tosi et al., 2022). Most attention involving pesticide impacts on pollinators have tended to focus on insecticides, with relatively less known about how herbicides, fungicides, and especially acaricides and their co-formulants, impact pollinator health (Straw et al., 2022). Research conducted on the different insecticide classes has also been skewed, with most recently a focus on neonicotinoids (a family of systemic insecticides), largely stimulated by prominent societal and political debate (Godfray et al., 2015; Pisa et al., 2015). In addition, published studies have tended to report mostly on lethal toxicity endpoints, with less information available on sublethal impacts of pesticide exposure (Gill and Raine, 2014; Siviter et al., 2018a; Siviter et al., 2018b; Tosi et al., 2022; Tosi and Nieh, 2019). It is also extremely important to understand the synergistic impacts of different pesticide classes, including insecticides, herbicides, and fungicides, as well as their co-formulants and 'inert' ingredients (Mullin, 2015; Straw et al., 2022), and how these interact with other stressors, such as pathogens (González-Varo et al., 2013; Grassl et al., 2018; O'Neal et al., 2018; Schwarz et al., 2022; Siviter et al., 2021; Tamburini et al., 2021a, 2021b; Tosi and Nieh, 2019; Tosi et al., 2022). Investigating these interactions is crucial for pollinator survival, and we are only now beginning to understand some of these complex dynamics. Even more scant, are studies examining the effect of different pesticide classes and their interaction with other factors, such as availability of flower/nutritional food resources (see Tosi et al., 2017; Klaus et al., 2021; Wintermantel et al., 2022). There is also little information on the long-term and cumulative effects impacting on different life-stages and castes (Tosi and Nieh, 2019).

In this paper, we first compare trends in the use of pesticides (insecticides, herbicides, fungicides) across different global regions from the 1995 to 2020 using the latest available global data from FAOSTAT[5] (countries with non-availability or inconsistency in data, either in one or both the years, have been excluded). We then use Automatic autoregressive integrated moving average (Auto ARIMA) models (Hyndman and Khandakar, 2008) to provide a forecast of likely trends in pesticide use (insecticides, herbicides, fungicides) in different global regions up to 2030.

We then highlight literature published since the 2016 IPBES pollination assessment that examines the effects of pesticides on diverse pollinator species. Literature since 2016 till 2022 was searched exhaustively in Scopus and Google Scholar using strings of relevant key words, and based on these analyses, we finish by discussing the future research trajectory and highlight a few global policy and conservation initiatives addressing pesticide use.

2. Trends of pesticide use across global regions

Pesticide use (insecticides, herbicides, and fungicides) has varied markedly across different countries, even within a global region, over the last two decades. The intensity of pesticide impact depends on application rate (kg/ha) and the toxic load of different pesticide formulations (Douglas et al., 2022). It is therefore prudent to consider both elements when judging the risk from pesticide use, particularly given the shift towards low application rate-high toxicity pesticides. However, for the purpose of this paper, we are limited to a focus on the application rate to understand variations in pesticide use since toxicity load assessments for important pollinator species are not available in most global regions. The maps portrayed in Fig. 1 compare these changes in application rates for insecticides (Fig. 1a), herbicides (Fig. 1b) and fungicides (Fig. 1c) over a span of twenty-five years i.e., 1995 to 2020.

Dark grey shading indicates no available data.

As reflected in the maps, between 1995 and 2020, there were visible increases in application rates of all three pesticide classes in South America and China. In Russia there has been an increase also. In South America, the increase can be attributed to rapid expansion of intensive farming in the last decades and either a lack of pesticide use regulation or its implementation (Zúñiga-Venegas et al., 2022). In China, the visible growth in pesticide use between 1995 and 2020 is due to strong state drive for agricultural industrialization (Xu et al., 2008). Russia saw steep falls in pesticide use (Oldfield, 2000) as the government withdrew support immediately after privatization of the collective farms following the dissolution of the Soviet state in the early 1990s (Mathijs and Swinnen, 1998). In the last two decades, changes to Russian state policy led to a rise in pesticide use (Zhilkin and Grigoryev, 2023).

To predict the future growth trends in pesticide use using time series data Autoregressive Integrated Moving Average (ARIMA) model (Hyndman and Khandekar 2008) was used. FAOSTAT[6] data was used for the analysis. ARIMA forecasting model is a combination of Autoregressive models and moving average models and is widely used for forecasting time series data (Awan and Aslam, 2020). We used the Automatic ARIMA version (Auto ARIMA) (Hyndman and Khandakar, 2008) for the analyses (Supplementary Note 1). The ARIMA time series forecast plots below show the future trends in pesticide use in different global regions (Fig. 2a - c). The best selected models and parameters justifying these selected models are provided in Supplementary Table 2.

2.1. Trends in insecticide use

Between 1995 and 2020, there has been a substantial increase in the total insecticide application rate across the world (Fig. 1a), but with substantial regional variation. An increase between 1000 and 3000 % was seen in countries such as Bhutan, Cambodia, Tajikistan, and Armenia in Asia; Mauritania, Nigeria, and Somalia in Africa; and Lithuania, Latvia, Austria, and Germany in Europe (Supplementary Table 1). In many other countries, insecticide use has grown over 100 % during this period (Supplementary Table 1). In 15 countries across different world regions, insecticide use has risen between 500 and 1000 %. They include Mozambique, Ghana, Botswana, Uganda, and Seychelles in Africa; Chile, Comoros and Ecuador in South America, Myanmar in Asia, and the Russian Federation in Europe (Supplementary Table 1). In yet another 28 countries, insecticide use has increased by 100 to 400 % (Supplementary Table 1). Pressure to intensify land productivity and greater availability of inexpensive insecticides are presumably some of the factors driving the rampant rise of insecticide use in most of these countries. In contrast, there has been a reduction in insecticide use in several developing and developed economies across different global regions, examples include India, Bhutan, Malaysia, Indonesia, Vietnam, United Kingdom, Norway, Australia, New Zealand and Cote d'Ivoire (Fig. 1a; Supplementary Table 1).

Insecticide application has increased between 1995 and 2020 and is expected to go up significantly in Africa and South America between 2020 and 2030 (Fig. 2a). All Asian regions except West Asia are expected to show a declining trend. In the rest of the regions evaluated, the present rate of insecticide use is predicted to remain stationary. It is noteworthy, however, that within different regions there are country-specific values of application rate that are greater than the regional median values (Supplementary Fig. 1a) attributable to country-specific factors, whereas the countries where a reduction is seen may be due to the introduction of insecticidal toxin producing GM crops (Brookes, 2020,

[5] Retrieved January 2023 from https://www.fao.org/faostat/en/#data

[6] Retrieved January 2023 from https://www.fao.org/faostat/en/#data

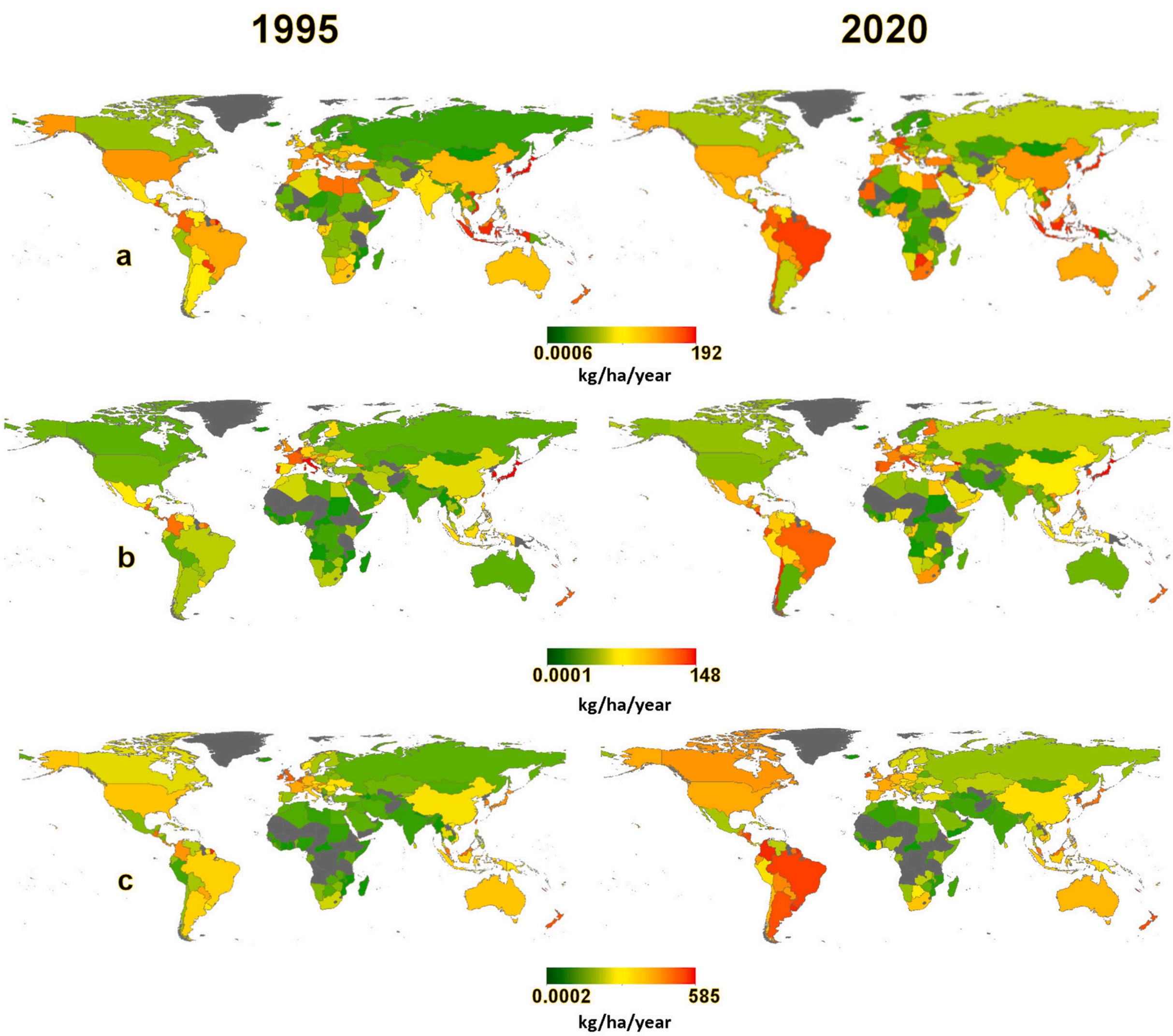


Fig. 1. Maps showing pesticide use in 1995 and 2020 for a) insecticides b) herbicides and c) fungicides. Colour gradients represent application rates (kg/ha per year) with warm colours indicating higher usage.

2022). The continued analysis and accurate mapping of insecticide trends rely on the accessibility of pesticide usage data (in general) at relevant scales, including at field-level (Douglas et al., 2022).

Furthermore, patterns in the use of specific insecticide families have not remained uniform during this period. While insecticide application has declined or is predicted to remain stationary till 2030 in Europe, there was significant growth in the use of neonicotinoids (until the EU 2018 ban[7] on their use in most outdoor situations). It must be noted that neonicotinoids are highly toxic to insects, so transition from other chemistries to neonicotinoids may reduce application rates but may greatly increase applied toxicity. For example, in the UK between 1994 and 2018, although use of some major insecticide families e.g., organophosphates, carbamates or pyrethroids either remained constant or declined, there was a marked rise in neonicotinoid use. However, a drastic drop in neonicotinoid use in 2020 reflected change in EU pesticide regulatory policy, much of which the UK retained even after its exit from the EU.[8] In India, however, although the proportion of neonicotinoid use in relation to total insecticide use has risen steadily over a five-year period (2016–2021) (Fig. 3b), older generation insecticides, like organophosphates, still constitute a major share of total insecticide use. Similar patterns may be found in other developing world regions too. A closer look at the total pesticide use data also reveals intra-regional variations. For example, in Africa the countries showing the largest increase in insecticide use include Gambia followed by Tanzania and Ghana, in Central America, Nicaragua and Costa Rica saw the largest increases, in South America, the largest increase was in Ecuador, followed by Bolivia, and in South Asia, the largest increase has been in Nepal, followed by Bangladesh (where there has been an 89 % increase in total insecticide use between 1995 and 2018, and particularly 118 % increase in pyrethroid use). In contrast, total insecticide use has fallen by 49 % in neighboring India. In South-East Asia, the largest increase has occurred in Myanmar, followed by Indonesia. In the European continent, use of insecticide has been highest in the Russian Federation.

However, it is important to note that the quantities of insecticides used is only indicative of the overall risk to non-target insects. For a thorough assessment, it would be important to consider the respective toxicities of different active ingredients and potential levels of exposure of different insect species (varying in their susceptibility) in different global regions (Mallinger et al., 2015; Uhl and Brühl, 2019; DiBartolomeis et al., 2019).

[7] Retrieved May 2023 from https://food.ec.europa.eu/plants/pesticides/approval-active-substances/renewal-approval/neonicotinoids_en

[8] Retrieved April 2023 from https://www.hse.gov.uk/pesticides/brexit.htm.

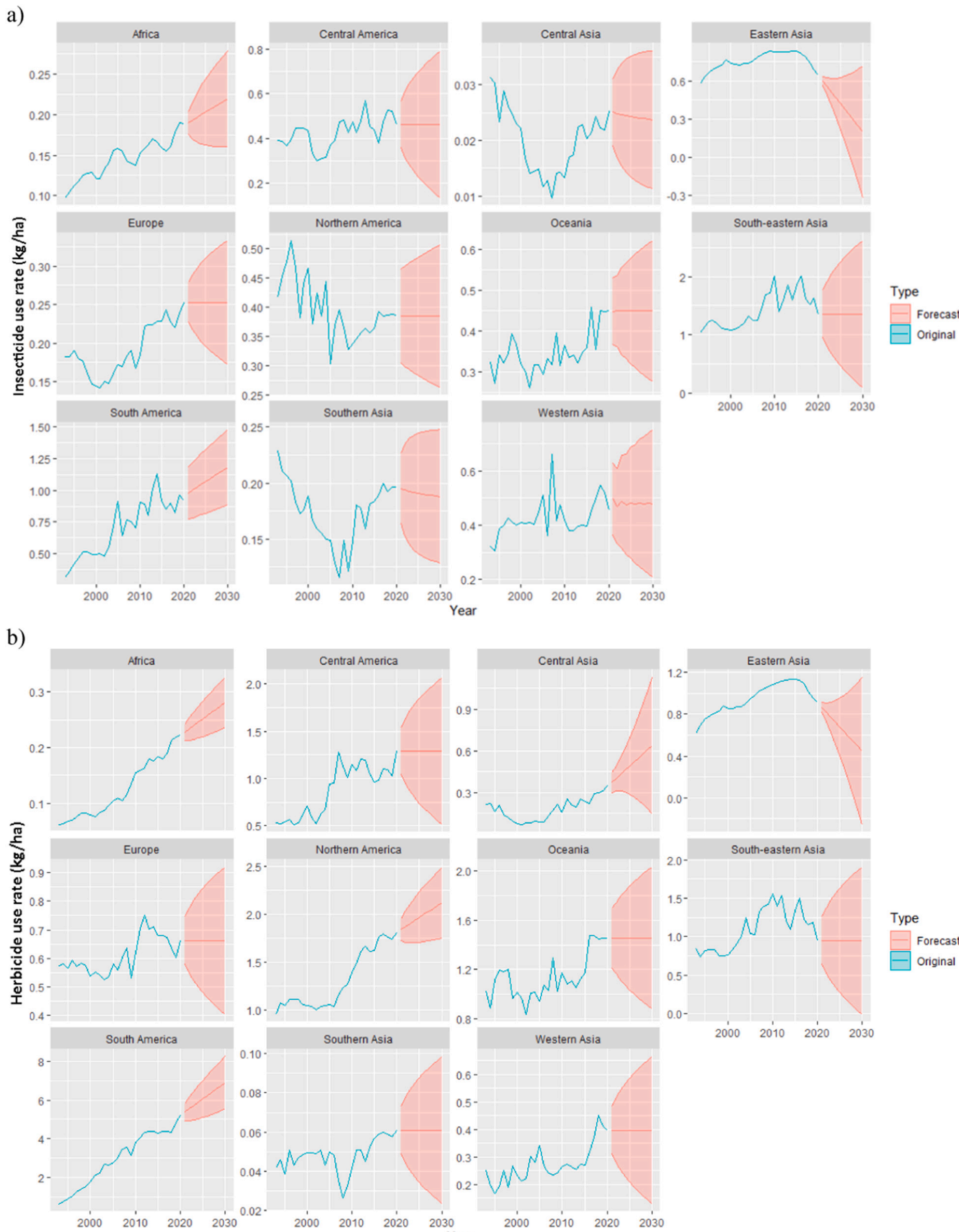


Fig. 2. Recent (1995–2020) and forecasted up to 2030 (best Auto ARIMA models) trends in pesticide use (kg/ha) in different global regions for a) insecticides, b) herbicides, c) fungicides.

Fig. 2. (*continued*).

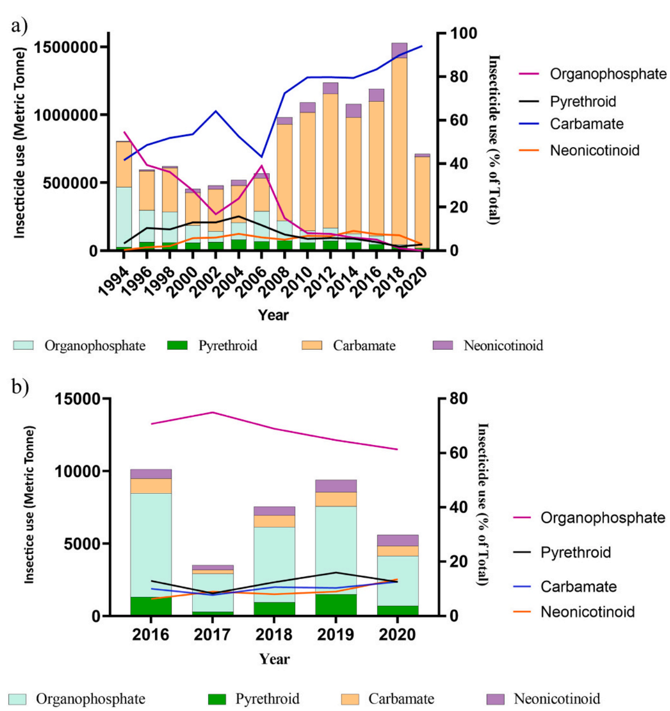


Fig. 3. Total and percentage use of different insecticides of all insecticides in a) UK and b) India up to 2020. Source a): Food and Environment Research Agency (FERA), Department for Environment, Food and Rural Affairs (Defra), Government of the United Kingdom); b) Source: Department of Agriculture & Farmers Welfare, Govt. of India.

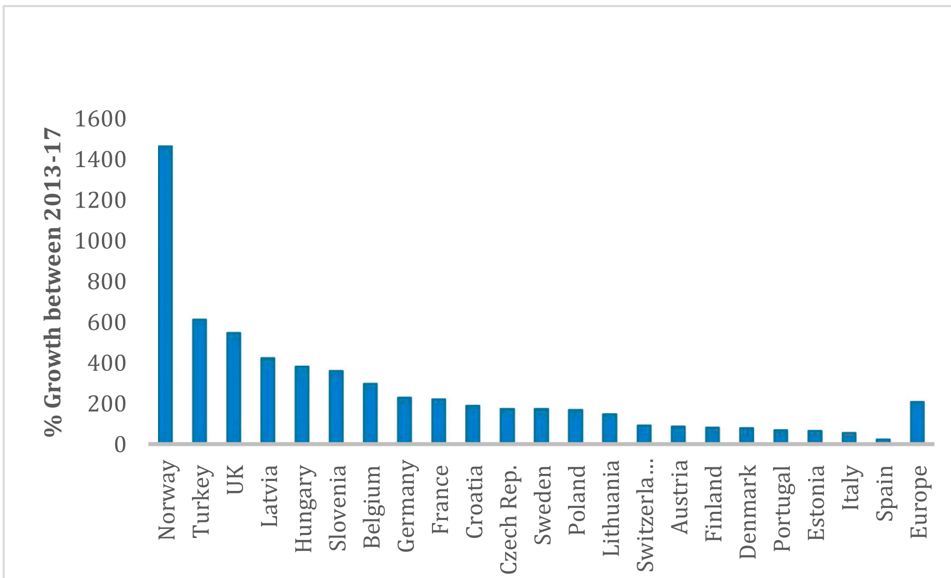
2.2. Trends in herbicide use

Herbicides account for 47.5 % of all pesticide usage worldwide (De et al., 2014; Grube et al., 2011; Sharma et al., 2019). Of various herbicides, glyphosate has been the most used across the world (Box 1). Its use in combination with genetically modified crops has further increased and expanded its application (Benbrook, 2012; Perry et al., 2016). However, as with insecticides, the application rate of herbicides across different global regions has not been uniform (Fig. 1b; Supplementary Table 1). The causes for this trend vary by region. For example, in South America there has been a wide-scale shift to no-tillage practices from 670,000 ha of no-till in the MERCOSUR countries (Brazil, Argentina, Paraguay, and Uruguay) in 1987 to over 30 million hectares in 2002. This increase in no-tillage practices combined with new and more efficient no-tillage seeding technology led to an increase in herbicide use to regulate weeds (Derpsch et al., 2008; Knapp and van der Heijden, 2018; Speratti et al., 2015). Countries like Honduras and Peru recorded over 1000 % growth in herbicide use and there have been over 500 % increase in Bolivia, Brazil, Ecuador and Uruguay (Supplementary Table 1). Outside of South America, other regions have also seen a substantial increase in herbicide use. In Africa there have been between 500 % to over 3000 % increase in Madagascar, Malawi, Mozambique, Seychelles, Somalia, Botswana, Gambia, Ghana, and Nigeria (Supplementary Table 1). Increases of similar magnitude in other Asian countries have been recorded in Tajikistan, Cambodia, Myanmar, Bangladesh, Maldives, Nepal, Armenia and Azerbaijan (Supplementary Table 1). Decline of agricultural labour in rural areas due to migration of people to cities has been cited as a reason for large scale adoption of herbicide in different parts of the world including in Asia and Africa (Gianessi and Williams, 2011; Hossain, 2015; Peterson et al., 2018). In North America, Canada showed over 200 % increase in herbicide use; in European countries, there has been an increase of between 100 % and 500 % in herbicide use that include Russia, Turkey, Spain, Serbia, Portugal, Albania, Estonia, Latvia, Lithuania and Poland (Supplementary Table 1).

Box 1
Glyphosate

Glyphosate is an organophosphorus systemic herbicide introduced first as Roundup(R) by Monsanto Inc. (Soares et al., 2021). Since its first commercialization in 1974, glyphosate- based herbicides (GBH) – sold under various brand names, has become one of the most sold herbicides in over 100 different countries (Benbrook, 2016) and its use has risen steadily. In the USA, glyphosate use in the agricultural sector rose 300-fold from 0.36 million kg in 1974 to 113.4 million kg in 2014 (Benbrook, 2016). In Europe, of the seven countries for which glyphosate use data is available between 2011 and 2017, France used the maximum amount allowable followed by Germany (Soares et al., 2021). However, between 2013 and 2017, Norway registered maximum use followed by Turkey and the UK (Antier et al., 2020b - ENDURE 2020 report[1]).

Figure Box 1. Percentage glyphosate growth in the EU countries between 2013 and 2017. Overall, 21 EU countries for which data was accessible registered a 209.7 % growth between 2013 and 2017.



Box 1. Data Source Antier et al., 2020a

[1] Retrieved January 2023 from https://doi.org/10.15454/A30K-D531

Growth in herbicide use in Canada has been potentially linked to the use of genetically modified corn, soya, and canola (Bacon et al., 2023). Historically, with non-GMO crops, farmers had to carefully time and control herbicide application to avoid harming crop productivity. However, the introduction of herbicide-tolerant GMOs, such as those resistant to glyphosate, allowed farmers to spray glyphosate-based herbicides (GBHs) directly on crops without damaging them. This shift led farmers to apply herbicides throughout the growing season, rather than just before planting or after harvest, significantly increasing overall herbicide use as they targeted weeds more frequently (Clapp, 2021). Since 1980, US total herbicide use continued to increase even though glyphosate use itself leveled off due to the glyphosate-alternatives being available (1980–2005). Although, after 2005, herbicide use increased significantly in the United States (Clapp, 2021). A major reason for this increasing trend in the United States has been the cultivation of genetically modified (GM) crops, a trend also seen in Canada (see above). In the USA, herbicide-resistant crop technology was associated with a 239 million kilogram (527 million pound) increase in herbicide use between 1996 and 2011 (Benbrook, 2012); the overall effect has been an increase in total pesticide use of around 183 million kgs (404 million pounds, c. 7 %) (ibid.).

Time series trends (1995–2020 forecasted up to 2030) show significant increasing trends in Africa, Central Asia, and South America. Eastern Asian countries on the other hand show a declining trend (Fig. 2b). Again, in several countries in a specific region, the use rate exceeds the regional median value (Supplementary Fig. 1b).

The status and trends of herbicide use should not be examined separately from the sharp rise in herbicide tolerant weeds – since both issues are tightly linked. Since 1991, 500 weed species have been recorded to be tolerant to herbicides worldwide (Supplementary Fig. 3). It is expected that this will lead to desperate and therefore increased use of more and more non-effective herbicides and also new products consequently increasing the vulnerability of pollinators.

2.3. Trends in fungicide use

Fungicide use has generally increased across different global regions but non-uniformly within specific regions. In several countries e.g., Yemen, Nepal, Bhutan, Bangladesh in Asia, Chile in South America, Mozambique, Somalia, Ghana, and Nigeria in Africa this growth has been of the magnitude of several thousand percent (Fig. 1c, Supplementary Table 1). In 30 countries, fungicide use increased between 500 and 1000 % (Supplementary Table 1), while another 38 countries saw increased growth between 100 and 500 %. The time series forecast models (Fig. 2c) based on existing data identify Africa, South America, South-East, East and West Asia as global regions that will see significant increase in fungicide use up to 2030.

3. Impacts of pesticides on pollinators

Pesticides can have both lethal and sublethal effects on pollinators, either directly or indirectly via effects on habitat, nesting and forage resources (IPBES, 2016). The following sections primarily cover information on the direct lethal and sublethal impacts on bees of insecticides, herbicides, and fungicides. Most of this reviewed evidence is from analyses published after the IPBES, 2016 assessment (albeit sometimes containing evidence from before that date but republished in data syntheses). The literature following 2016 up to 2022 was exhaustively searched using relevant search strings in Scopus and Google Scholar.

3.1. Insecticides

3.1.1. Lethal toxicity

Studies and syntheses reporting lethal impacts on bees (Delkash-Roudsari et al., 2020; Douglas et al., 2020; Lunardi et al., 2017; Lundin et al., 2015; Nowakowski and Pywell, 2016; Tosi et al., 2022; Yao et al., 2018) have predominantly focused (and continue to do so) on the managed western honey bee *Apis mellifera* (e.g., US EPA Ecotox database[9]), and only on a restricted number of insecticides (IPBES, 2016). However, there have been recent moves to assess lethal endpoints (e.g., LD5O, % mortality and survival) for other bee species, including Asian honey bee *A. cerana* and other solitary bees (e.g., Megachilids) and stingless bee species (e.g., Meliponini) (Barbosa et al., 2015; Dorneles et al., 2017; Jacob et al., 2019a, 2019b; Ma et al., 2022; Soares et al., 2015; Otesbelgue et al., 2018; Padilha et al., 2020). During 2016–2022 there have been increasing numbers of studies on *A. cerana*. For instance, cytochrome P450 monooxygenases (P450s) are known to be instrumental in metabolic detoxification of insecticides (Lu et al., 2021). A P450 mediated detoxification system has been found in *A. mellifera* (Berenbaum and Johnson, 2015; du Rand et al., 2015). It has recently been reported that like its European congener counterpart, *A. cerana* also has a P450 enzyme mediated detoxification system, and at least four regulatory genes have also been identified (Zhang et al., 2019). There are findings, however, to suggest that *A. cerana* might be more sensitive to neonicotinoid-mediated acute toxicity (estimated by LD50) compared to *A. mellifera* (Li et al., 2017a; Yasuda et al., 2017). Li et al. (2017b) attribute this to differences in innate immune responses of the two species. It has also been suggested that the differences in sensitivity of the two species may vary according to the structure of different neonicotinoid insecticides, but not with body mass of bees (Yue et al., 2018). A more recent study (Linguadoca et al., 2022) has shown intraspecific variation in sensitivity to different families of insecticides and although body weight partially explained these variations it is possible that sensitivity may vary according to sex and caste.

Studies on the Megachilids are rather skewed towards *Osmia* spp. and *Megachile* spp. (Fig. 4, Supplementary Table 3) (Boff et al., 2021; Centrella et al., 2020; Claus et al., 2021; Fortuin et al., 2021; Kopit et al., 2022; Mokkapati et al., 2021; Song et al., 2021a; Song et al., 2021b; Stuligross and Williams, 2020; Stuligross and Williams, 2021; Tomé et al., 2017). However, the reports mostly deal with neonicotinoids, and lethal endpoint studies for the older generation insecticides (e.g., organophosphates or organochlorines) are still lacking despite their ongoing large-scale use in many developing regions.

The cumulative number of studies reporting lethal endpoints for stingless bees has also grown since the IPBES report (2016) (Brigante et al., 2021), with lethal endpoints for nine species of stingless bees reported between 2016 and 2022 from Brazil alone (Supplementary Table 3). This is an important development since the stingless bees (Meliponines) take about twice as long to reach worker adulthood from egg compared to the honey bees, which makes stingless bee broods potentially more vulnerable to insecticide exposure (Cham et al., 2018). This makes risk assessment standards for the Meliponine bees based on the life cycle of the honey bees less accurate. More information coming out on the stingless bees is therefore most useful for evolving their risk assessment standard.

There remains a need in many global regions to move pesticide risk assessment beyond *A. mellifera* as a single focal species considering the significant life-history trait differences among various bee species. This is supported by reports of lethal impacts on bumble bees in temperate regions (Banks et al., 2020; Ellis et al., 2017; Linguadoca et al., 2022; Minnameyer et al., 2021; Mobley and Gegear, 2018) and stingless bees of the tropics (Bernardes et al., 2018; Brigante et al., 2021; Dorneles et al., 2017; De Oliveira Ferreira et al., 2020; Piovesan et al., 2020). While the breadth of evidence on lethal impacts on multiple bee species is growing, there remains a general lack of knowledge of lethal intra-specific pesticide impacts on different sexes, castes, and life-stages of the test species. Moreover, most risk assessments used by regulators (and sometimes by researchers) are conducted over short time periods (e.g., 10 days) and so fail to assess the potential for cumulative impacts resulting in lethal effects (Simon-Delso et al., 2018).

The lethal risk to bees from the full array of available pesticides has also not been fully considered. Notably, the so-called "safer" insecticides of either natural origin (e.g., plant or soil bacterium-based insecticides such as Azadirachtin, Spinosad, Avermectin) or the growth regulators (e.g., Novaluron) have equally proven to be harmful to solitary and stingless bees (Supplementary Fig. 2). Out of 12 studies (Supplementary Table 4) published between 2016 and 2022 that reported impacts of a natural origin insecticide, eight have reported lethal non-target toxicity for such insecticides (Araújo et al., 2019a, 2019b; Brigante et al., 2021; De Oliveira Ferreira et al., 2020; Gómez-Escobar et al., 2018, Marques et al., 2020; Piovesan et al., 2020; Tomé et al., 2015b). Further work is needed to understand the contribution to lethal impacts (direct and synergistic) in bees of so called 'inert ingredients' in insecticides added as co-formulants along with the active ingredients (Straw et al., 2022).

3.1.2. Sublethal toxicity

A number of important studies on the sublethal impact have come out during 2016 and 2022 on various species including on previously less studied species like the Asian honey bee *Apis cerana* (adults and brood) and other species from regions outside the Western world. Many of these studies include impacts of non-neonicotinoid insecticides e.g., on olfaction, on microbiome, p450 mediated detoxification, visual acuity, and homing (Li et al., 2017b; Ma et al., 2019; Tan et al., 2017; Yang et al., 2019; Zhang et al., 2019).

A few studies have shown that insecticide mixtures, including co-formulants, used in intensively managed agricultural landscapes can adversely affect the *A. cerana* olfaction (Chakrabarti et al., 2015) and vision (Chakrabarti et al., 2019) in the field. Kumar et al. (2022) found increased apoptotic cell death in the brain tissue could be responsible for these effects. Further information about effects of neonicotinoid insecticides on homing behaviour has also emerged (Tosi et al., 2017).

Several recent studies highlight sublethal behavioural impacts on bumble bees (*Bombus* spp.) (Crall et al., 2019; Lämsä et al., 2018; Leza et al., 2018; Muth and Leonard, 2019; Minnameyer et al., 2021; Siviter et al., 2021; Stanley et al., 2016; Tasman et al., 2020; Crall et al., 2018; Raine and Rundlöf, 2024). Behavioural impacts include sleep disorder, foraging demotivation, sub-optimal foraging decisions and within-colony nursing behaviour (Chole et al., 2022; Lämsä et al., 2018; Siviter et al., 2021; Tasman et al., 2020).

Several studies also report sublethal physiological impacts of neonicotinoids on bumble bees. Such physiological impacts include thermoregulation (Crall et al., 2018), sperm gland physiology (Minnameyer et al., 2021) and nutritional distress (Linguadoca et al., 2022). However, a number of studies have also reported no significant impacts of different insecticides including neonicotinoids on *A. mellifera* or *Bombus terrestris*

[9] Retrieved January 2024 from https://cfpub.epa.gov/ecotox/

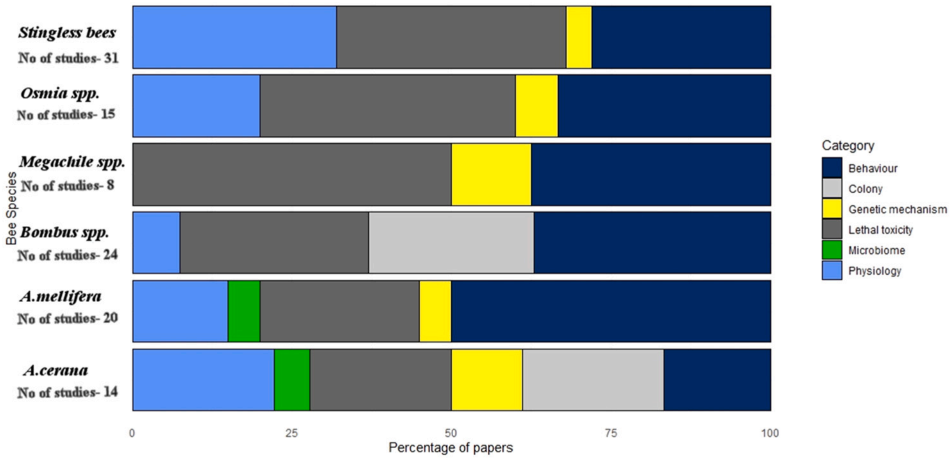


Fig. 4. Summary of publications from 2016 to 2022 reporting different lethal and sublethal impacts of insecticides for different pollinator species or groups. The total number of publications focusing on different bee taxa are reported, as well as the percentages of publications within each taxa focusing on different impacts.

(Osterman et al., 2019; Schwarz et al., 2022; Siviter et al., 2019; Straw and Brown, 2021b).

Lehmann and Camp (2021) provide a systematic review of the effects of pesticide exposure on solitary bees with a rise in the number of publications over the last decade, which peaked post-2015. Although the review deals with only a few species of commercially available solitary bees (*Megachile rotundata, Nomia melanderi*, four species of *Osmia* and *Eucera pruinose*), it considers several studies on impacts of organophosphates, carbamates, and neonicotinoids. Most of these studies involve adults, with few or no studies on immature life stages. Comparison of the sub-lethal sensitivities of different solitary species to different pesticide classes remains a knowledge gap, although Ansell et al. (2021) found *M. rotundata* to be more sensitive to a neonicotinoid compared to organophosphate and pyrethroid. *M. rotundata* lacks the P450 enzyme detoxification system (Hayward et al., 2019) found in *Apis*, with implications for future regulatory guidelines of a pesticide risk assessment framework. In a major systematic review and meta-analysis, Siviter et al. (2018c) found significant negative effects of insecticides on learning and memory (i) at field realistic dosages, (ii) under both chronic and acute application, and (iii) for both neonicotinoid and non-neonicotinoid insecticide groups. Although the impacts of neonicotinoids are well documented, the mechanistic framework of the possible mode of their action is still being understood. Pamminger et al. (2018) suggested that there might be a close ontogenic association between the haemocytes of the insect immune system and the nervous systems and that this connection makes the immune system of pollinators and other insects inherently susceptible to interference by neurotoxins such as neonicotinoids at sublethal doses. Raine and Rundlöf (2024) review pesticide exposure, exposure pathways and impacts (lethal and sublethal) on non-*Apis* bees since honey bees and other bees differ in biology, foraging and nesting behaviour and degree of sociality; this review highlights those environmental risk assessments (ERAs) that did not consider non-*Apis* bees. Raine and Rundlöf (2024) point out there are other regulation and policy tools beyond ERAs that can consider the pesticide threat to bees.

Insecticides of natural origins have been reported to inflict sublethal effects also on several stingless and solitary bee species (Supplementary Table 4). For some stingless bees, these insecticides have been shown to be as harmful as neonicotinoids in producing aberrations in learning, development, or physiology (Bernardes et al., 2017, 2018; Marques et al., 2020; Piovesan et al., 2020; Tomé et al., 2015a). Challa et al. (2019) reported impacts of a few widely-used biopesticides, e.g., Azadirachtin, Anonnin, Beauveria bassiana and Bt var. on Asiatic honey bees, *Apis cerana*, and found them to impair foraging rate and foraging speed.

3.2. Herbicides

Herbicides constitute almost half of the total global volume of pesticides used (Sharma et al., 2019). Herbicides can indirectly affect the bees' nutrition and survival by decreasing the non-crop forage resources both in the crop and in adjacent non-cultivated habitats (Belsky and Joshi, 2020; Bohnenblust et al., 2016; Jacobson et al., 2018). However, in comparison to insecticides, the direct lethal or sublethal impact of herbicides on the pollinators have been less studied, though in recent years, papers on herbicide impacts are increasing. Our literature search revealed 22 studies between 2017 and 2021 that showed sublethal and lethal impacts of herbicides on bees (Fig. 5, Supplementary Table 5). Herbicides - either used as a single chemical or as multiple herbicide mixture - have been reported to be directly harmful in 75.0 and 88.9 % studies (Iwasaki et al., 2020). Between 2016 and 2022, 24 studies were published including four comprehensive reviews (Battisti et al., 2021; Belsky and Joshi, 2020; Cullen et al., 2019; Iwasaki et al., 2020). More than half of the studies are from South America (Brazil, Argentina, and Colombia) while six are from North America (USA, Canada), three from China and a single study from Africa (Ghana). All these studies reported lethal and sublethal effects of herbicides (Fig. 5, Supplementary Table 5); mostly involving glyphosate, which a meta-analysis involving 17 studies and 34 data sets on *A. mellifera*, and a few stingless bee species showed mortality in bees (Battisti et al., 2021).

3.2.1. Lethal toxicity

One of the better studied herbicides is glyphosate, exposure to which has been shown to negatively affect both larval and adult *A. mellifera* survival (Battisti et al., 2021; Chaves et al., 2021; Dai et al., 2018; Faita et al., 2020; Jumarie et al., 2017; Mengoni Goñalons and Farina, 2018; Motta et al., 2020; Odemer et al., 2020; Tomé et al., 2020; Vázquez et al., 2018). Glyphosate has been found to be lethal for *Melipona quadrifasciata* at the colony level as it kills all larvae within a few days (Seide et al., 2018). Glyphosate has been reported to impact gut microbiota and survival of the honey bees (Motta et al., 2020). A recent study has however suggested that it is the surfactant, and not the active ingredient glyphosate, that causes mortality in bumble bees (Straw et al., 2021), further highlighting the need to focus research on disentangling the toxicity of the different active ingredients and co-formulants of pesticides and their combinations.

3.2.2. Sublethal toxicity

Farina et al. (2019) reports various sublethal adverse effects of glyphosate on *A. mellifera*, including associative learning processes of foragers, cognitive and sensory abilities of young hive bees and delayed

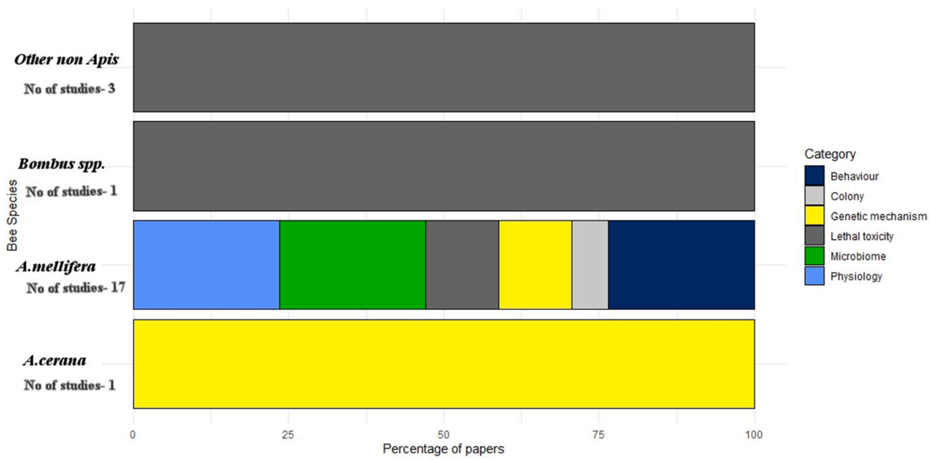


Fig. 5. Summary of publications from 2016 to 2022 reporting different lethal and sublethal impacts of herbicides for different pollinator species or groups. The total number of publications focusing on different bee taxa are reported, as well as the percentages of publications within each taxa focusing on different impacts.

brood development. Vázquez et al. (2020) reported *A. mellifera* foragers showing reduced body and muscle movement including antennal activity from exposure to as little as 50 ng of glyphosate, with implications for foraging orientation capacity. During 2016 and 2022, more studies on non-*Apis* bees have further underscored the adverse sub-lethal direct impacts of glyphosate (Araújo et al., 2021; Graffigna et al., 2021; Nocelli et al., 2019; Seide et al., 2018; Straw et al., 2021). A few studies have also reported adverse effects of glyphosate on the bee gut microbiome, increasing potential vulnerability to pathogen infection (e.g., increased worker bee mortality subsequently exposed to the pathogen *Serratia marcescens* (Motta et al., 2018)). Glyphosate has been shown to negatively impact bumble bees thermoregulation (Weidenmüller et al., 2022).

3.3. Fungicides

Fungicides thought to be safe for pollinators are now increasingly seen to cause both lethal and sublethal effects on bees (Fisher et al., 2021a, 2021b, 2021c). Our search revealed 23 studies published between 2016 and 2022 (Fig. 6, Supplementary Table 6) that reported lethal toxicity and sublethal impacts on physiology and behaviour. Single fungicide chemicals can be directly harmful to bees by altering metabolism, reproduction, and food consumption (Mao et al., 2017) or indirectly by increasing insecticide toxicity (Sgolastra et al., 2017; Tsvetkov et al., 2017). Eleven studies in in the years encompassing our study (2016–2022) have also shown synergistic effects of fungicides used in combination (Supplementary Table 6).

3.3.1. Lethal toxicity

Several studies and reviews in during 2016–2022 have found that fungicides alone can affect bee survival for both larval and adult stages (Fisher et al., 2021a; Fisher et al., 2021b, 2021c; Jaffe et al., 2019; Tamburini et al., 2021a; Wu et al., 2022) (Fig. 6, Supplementary Table 6). Various fungicides that are responsible for causing lethal toxicity include Difenoconazole (Almasri et al., 2021; Leite et al., 2022), Boscalid, and Pyraclostrobin combination (Fisher et al., 2021a, 2021b), Amistar (Straw and Brown, 2021a) and Propiconazole (Han et al., 2019).

3.3.2. Sublethal toxicity

Fungicides used in combination with other insecticides have been found to interfere with the detoxifying mechanisms in bumble bees (Raimets et al., 2018), but this was not the case in other studies (Schwarz et al., 2022; Tamburini et al., 2021a, 2021b). Non-*Apis* bees e.g., *Osmia* spp. and the stingless bees have been shown to be more sensitive to the synergistic actions of fungicides than *A. mellifera* (Azpiazu et al., 2021; Brigante et al., 2021; da Costa Domingues et al., 2020; Iverson et al., 2019; Sgolastra et al., 2017 but see Schwarz et al., 2022; Wade et al., 2019; Wernecke et al., 2019). Fungicide used as a co-formulant has been shown to have physiological impacts (Straw and Brown, 2021a).

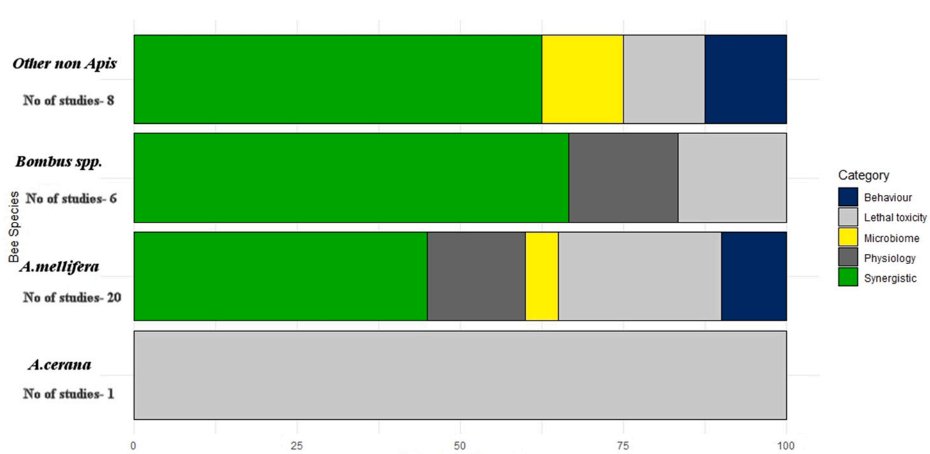


Fig. 6. Summary of publications from 2016 to 2022 reporting different lethal and sublethal impacts of insecticides for different pollinator species or groups. The total number of publications focusing on different bee taxa are reported, as well as the percentages of publications within each taxa focusing on different impacts.

However, Robinson et al. (2017) reported no impact of fungicide on *A. mellifera, Bombus terrestris* and *Osmia bicornis.*

4. Advancement of knowledge and existing gaps

Our understanding of the effects of pesticides on pollinators, particularly bees, was the focus of our review. New information is emerging from other global regions beyond North America and Europe, as well as on previously under-researched species of pollinators. However, the overall geographic skew in information towards the developed economies remains and fewer pesticide impact studies on pollinators are being done in major world regions where pesticide use has been undergoing dramatic changes. To compound the challenges faced with pesticide use and potential impacts on pollinators, in low and lower-middle income countries there is a lack of information and data on pesticide application rates leading to underestimation of actual usage (Shattuck et al., 2023). Even in high income countries, such as the United States, pesticide use data used to include pesticides applied as seed treatments, but this is no longer the case (Hitaj et al., 2020).

Fig. 7 a - c illustrates the extent of differences in insecticide, herbicide, and fungicide usages in different countries between 1995 and 2020 overlayed with the number of studies available for different regions. Russia, Australia, Canada, Central and West Asian countries, and several countries in Africa have experienced many fold increases in insecticide use over the last twenty-five years, but with almost no studies on pollinator impacts in recent years till 2022. However, many studies have appeared from countries with large increases in insecticide use e.g., Brazil, USA, and China since 2016. Most South American countries, USA and Canada, Russian Federation, several African countries, and countries in the Central and West Asia regions have experienced large scale growth in herbicide use. Although relatively fewer in number compared to insecticide impact studies, most studies on herbicide impacts are again from South America (Brazil and Argentina) and the USA. There is, however, an overall lack of studies from most areas with substantial herbicide usage growth around the world. For fungicides too, there is no information from most world regions with substantial growth in fungicide usage.

Focus of studies on different bee taxa also vary across different global regions (Fig. 8a – c). For example, insecticide, herbicide, and fungicide impact studies from South America are predominantly on Meliponine bees while studies on Megachilid bees are mostly from North America. Studies on *Osmia* sp. are predominantly from Europe. Studies on *A. cerana* are expectedly from Eastern Asia. Studies on Bumble bees are major focus in Europe.

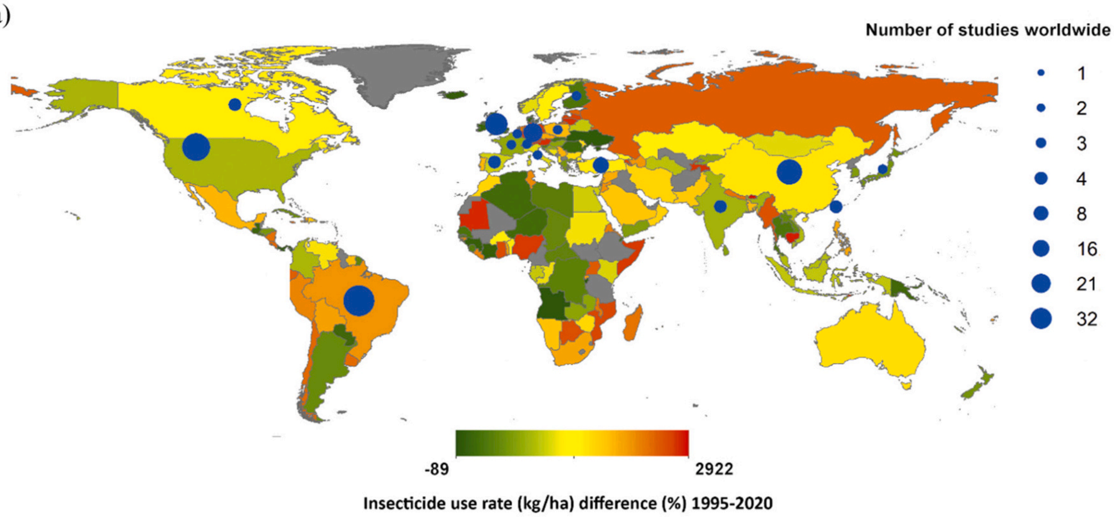
Information on effects of diverse groups of insecticides other than the neonicotinoids are also emerging. Comparative studies on effects of a specific pesticide on different bee species, different pesticides on several species, or effects of a single chemical or in synergy with other chemicals including co-formulants have emerged (Siviter et al., 2021). The “inactive” or “inert” pesticide ingredients (formulation ingredients or adjuvant components) can also have a negative impact on pollinators and are currently understudied in most lab and field tests (Mullin, 2015; Straw et al., 2022). There are some key insights, too, that should inform future risk assessment frameworks; that the so-called ‘safer insecticides’ of natural origin can have similar adverse impacts as neonicotinoids is revealing (e.g., Bernardes et al., 2017, 2018). Equally notable are the adverse impacts on bees of some of the new generation of insecticides e.g., Sulfoxaflor (Siviter et al., 2019, 2020). The vulnerability of solitary species of the Megachilidae family owing to the lack of the detoxifying P450 enzyme system that both *A. mellifera* and *A. cerana* possess is also alarming (Haas et al., 2022). However, information on Megachilids is almost exclusively from North America or UK/Europe, and nothing is known about these species in other parts of the world. Similarly, there is no information on any species from the entire Halictidae, Colletidae and Andrenidae families. Another large gap in our understanding is that of pesticide impacts on non-bee pollinators e.g., Dipterans and Lepidopterans, which are well known to be important crop and wild flower pollinators (IPBES, 2016; Rader et al., 2020). Research conducted on these groups tends to be from a pest control point of view, and on how effective pesticides are in dealing with them. There are some exceptions, for example investigations into whether herbicide impacted Monarch butterflies in the USA (see Agrawal, 2019 for a brief synthesis). There is also a little to no information on insecticide impacts at the level of ecological communities, and how these may influence ecological networks. We need to understand how insecticides affect pollinator communities and their complex interactions in food webs at different scales and at field realistic exposures. In reality this may involve understanding how a mix of different pesticides impact bees in interaction with other global change drivers such as climate change (Dirilgen et al., 2023).

Reports on herbicide impacts are mostly from the Global North. Even though information is now emerging from a wider geographic region (Supplementary Table 5), there remains a general lack of data from Asia and Oceania. Information on herbicides is also still heavily biased towards effects on *A. mellifera* and up to 66 % of studies in one review focused mostly on glyphosate (Dirilgen et al., 2023; Cullen et al., 2019). Future research should disentangle the toxic effects of individual and combinations of active ingredients as a recent review showed that for 140 active ingredients examined, data of impacts on bees are still generally lacking (Iwasaki and Hogendoorn, 2021). Finally, as a recent systematic review has highlighted, we also need to understand a great deal more about how different fungicides cause harm to different pollinator species in synergy with active ingredients of insecticides and herbicides (Dirilgen et al., 2023).

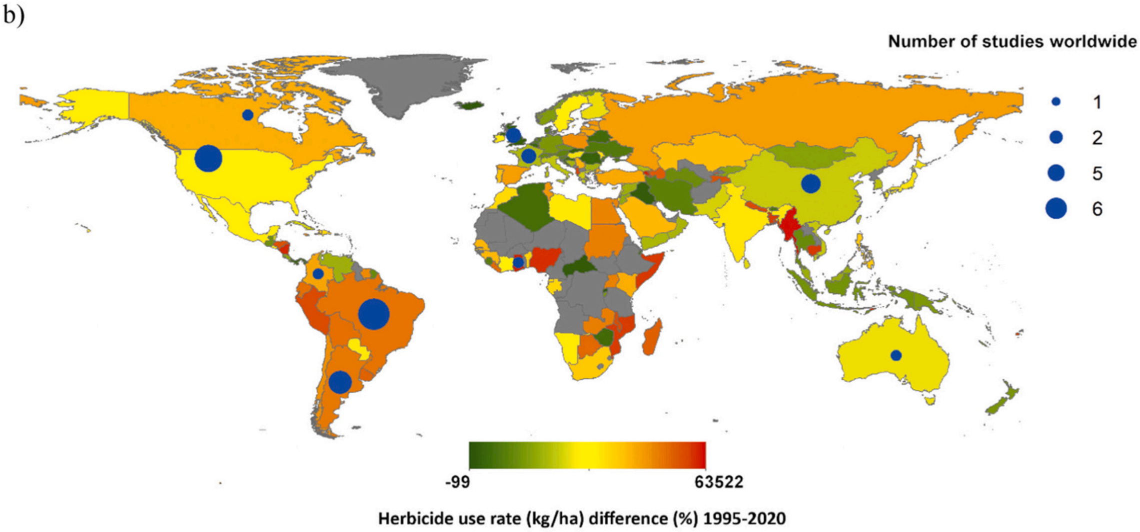
5. Towards a comprehensive pesticide risk assessment framework

In a first ever global analysis of pesticide pollution risk, with 92 active pesticide ingredients across 168 countries, Tang et al. (2021) found that 31 % of countries with agricultural land are at high risk from pesticide pollution and of those countries, 34 % are found in biodiversity rich areas. The most vulnerable areas include South Africa, China, India, Australia, and Argentina. A key outcome of our analysis shows pesticide use has been growing in these regions too. A number of regions where pesticide use has been increasing overlap with regions of high to moderately-high bee diversity (Orr et al., 2021), therefore, increasing the potential risk to bee pollinators. Regulatory regimes vary greatly across global regions and the majority of toxicity assessment protocols with respect to pollinators are conducted using *Apis mellifera* and do not include assays of sublethal effects, although solitary bees are now included in European risk assessment protocols (European Food Safety Authority, 2013). Different bee species have different vulnerabilities to pesticides (and their combinations) because of their different life-history and ecological traits (described above in Section 3); therefore, it is important to broaden our scientific knowledge on the risk profile for different taxonomic and functional groups of pollinators (Raine and Rundlöf, 2024). A deeper knowledge base can then inform the development of a more inclusive risk assessment framework that builds on the European example (Uhl and Brühl, 2019; European Food Safety Authority, 2013).

Importantly, assessment frameworks essentially also do not include ecosystem level studies. For example, few consider the ecological context and landscape heterogeneity in which pollinators are exposed to pesticides (e.g., multicounty comparisons of field-realistic exposure to neonicotinoid-treated oilseed rape crops - Woodcock et al., 2017). Because of a lack of different foraging resources, pollinators in simple landscapes dominated by few plant species may well experience poorer nutrition and be less resilient to pesticide effects than those existing within more plant diverse systems (Crone et al., 2022). However, while there is potential for semi-natural habitat or flower plantings to dilute the risk to foraging pollinators of exposure to pesticides, the evidence is not yet conclusive (Obregon et al., 2021; Park et al., 2015; Rundlöf et al., a)



b)



c)

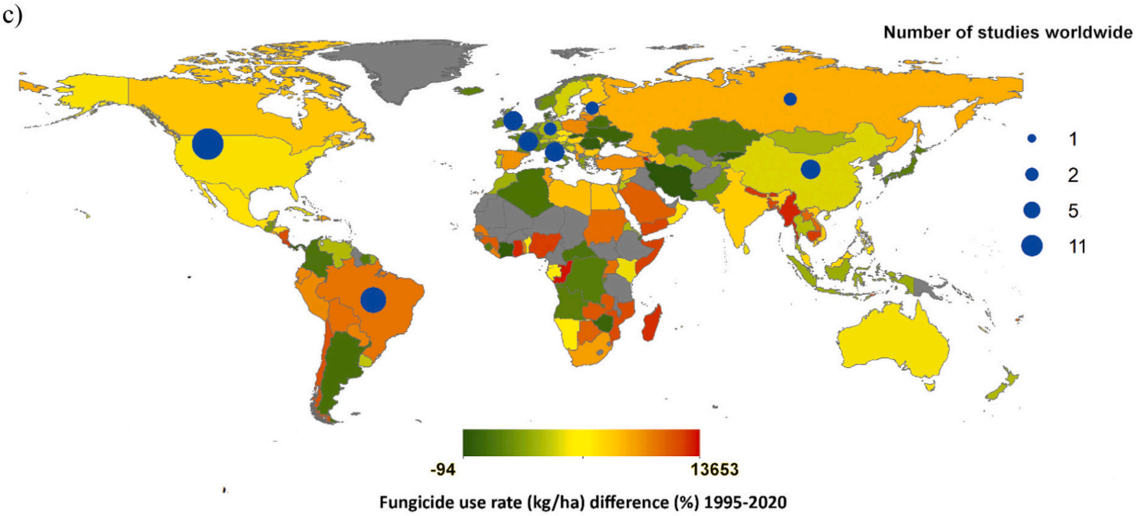


Fig. 7. Differences (% change) in use rates (kg/ha) between 1995 and 2020 for a) insecticides, b) herbicides and c) fungicides. Circles on the maps indicate the number of papers (scaled by the number) published during 2016 and 2022.

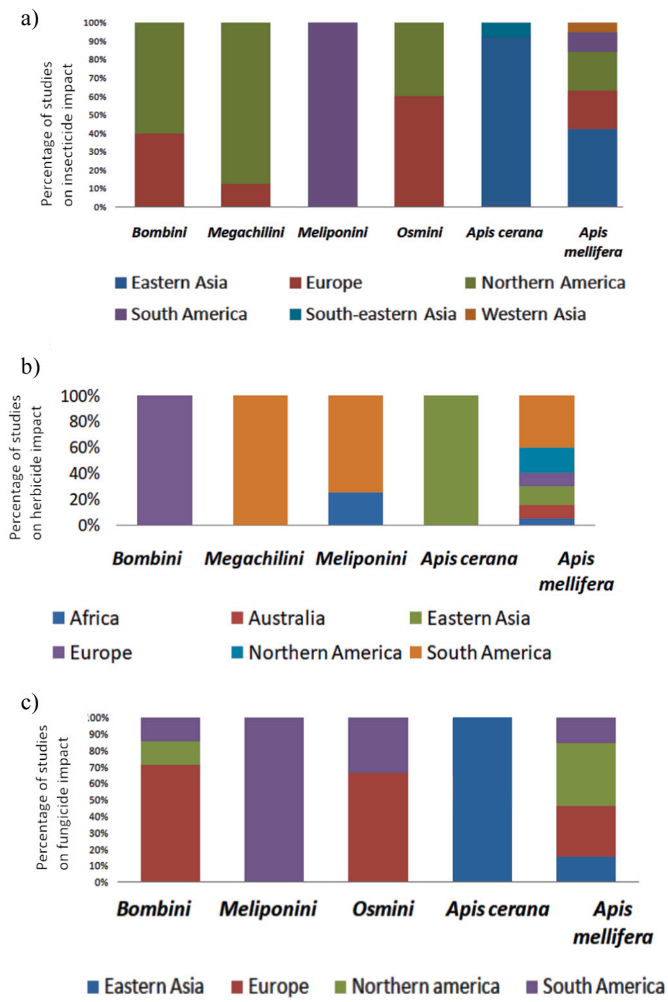


Fig. 8. Percentage of studies on impact of a) insecticide, b) herbicide and c) fungicide for different bee taxa across different global regions during 2016–2022.

2022). Further research is needed in this area, such as the optimal amount or spatio-temporal configurations of semi-natural to cropped habitats in order to diminish pesticide exposure risks and support pollinator nutritional resources.

6. An outlook on pesticide policies

Various policies in the last decade have focused on reducing (i.e., involving appropriate use of pesticide application) or eliminating the use of pesticides. Existing efforts to monitoring and track pesticide use, such as the US Geological Survey's National Pesticide Use Maps (USGS) are in danger of being limited or reduced due to budgetary constraints (Gewin, 2023). Policies or legislation that reduce or fully discontinue the use of pesticides should provide users with safe and effective alternative pest control strategies (Jactel et al., 2019) – which may not currently exist or accessible information about these alternatives may be limited. Moreover, current pesticide risk assessments, which are defined by metrics such as LC50 or LD50, and guidance on recommended use, can be disconnected from real-world situations, such as ecological contexts and management legacy effects. This disconnect can create ambiguity for decision-making or policymaking (Siviter et al., 2023). An example is the use of alternative "safer" pest control strategies being complicated by the detection of residues from banned pesticides (and their potential impacts) long after their use has stopped (Siviter et al., 2023; Wintermantel et al., 2020). There are many other strategies outside of "safe" pesticides which includes integrated pest and pollinator management (IPPM) - promoting insecticide treatments with low bee toxicity (Leach et al., 2022). More detailed discussions regarding pollinators, pollination services and pesticide policies have been reviewed elsewhere (Dicks et al., 2016; Gemmill-Herren et al., 2021; Hipólito et al., 2021; Galetto et al., 2022). Below we provide some illustrative examples of regional and global efforts towards safeguarding pollinators from pesticide use.

A new deal for pollinators: a revision of the EU Pollinator Initiative.

In 2013, Europe restricted the use of three neonicotinoids (clothianidin, imidacloprid and thiamethoxam) in seed coating in order to protect wild pollinators and managed honey bees foraging on flowering crops. In 2018, the European Union then extended this to a ban on these three neonicotinoids. The European Commission released the initial EU Pollinator Initiative in 2018 and more recently, a communication from the European Commission (January 2023)[10] on the revision of the EU Pollinator Initiative. The second objective of the revised EU Pollinator Initiative is: Improving pollinator conservation and tackling the causes of their decline – the protection of pollinators from pesticides falls under this objective. The specific target is to "reduce the risk and use of pesticides and the use of more hazardous pesticides by 50% by 2030" explicitly written in the EU Farm to Fork Strategy[11] and the EU Biodiversity Strategy for 2030.[12] In addition, the Nature Restoration Law was formally adopted by the European Council (June 2024), and Article 10 of this regulation has specific text to put "in place in a timely manner appropriate and effective measures" to reverse the decline of pollinator populations by 2030. Additional text on specifies "…prohibiting the use of pesticides in ecologically sensitive areas[13]…".

6.1. *International Code of Conduct on the Distribution and Use of Pesticides*

The International Code of Conduct on the Distribution and Use of Pesticides (hereinafter referred to as the "Code of Conduct"), is a guidance document developed jointly by FAO and the World Health Organization (WHO) and adopted by the FAO Council at the twenty-fifth session of the FAO conference (1985)[14] and was established to attempt to standardize pesticide use and distribution across countries (WHO/FAO, 2014). In addition, the Code of Conduct also develops guidelines for pesticide management, registration, risk assessment, use and applications, and waste disposal among others (WHO/FAO, 2014). Although this instrument is non-legally binding (i.e., a voluntary framework for governments and other stakeholders), countries are "strongly encouraged to adopt the standards set out in the Code of Conduct". In the last survey results of countries (2018), pesticide legislation has been established in 53 of 56 responding countries, but one-third of the responding countries lack national guidelines on the pesticide registration process. Given these results the FAO Pesticide Registration Toolkit[15] was developed. In terms of management, policies on Integrated Pesticide Management exist in 35 of 51 responding countries and two-thirds of responding countries report "major problems with pesticide resistance

[10] Retrieved January 2023 from https://environment.ec.europa.eu/publications/new-deal-pollinators_en

[11] Retrieved February 2023 from https://food.ec.europa.eu/horizontal-topics/farm-fork-strategy_en

[12] Retrieved February 2023 from https://environment.ec.europa.eu/strategy/biodiversity-strategy-2030_en#:~:text=The%20EU's%20biodiversity%20strategy%20for,contains%20specific%20actions%20and%20commitments.

[13] Retrieved April 2024 from https://www.europarl.europa.eu/doceo/document/TA-9-2024-0089_EN.html

[14] Retrieved January 2023 from A revised version was adopted by the Hundred and Twenty-third Session of the FAO Council (2002). https://www.fao.org/3/y4544e/y4544e00.htm

[15] Retrieved April 2023 from https://www.fao.org/pesticide-registration-toolkit/en/

in agriculture" (World Health Organization and Food and Agriculture Organization of the United Nations, 2019).

6.2. International Pollinator Initiative

The Code of Conduct (see above) was not the only FAO-led global initiative that involves pesticide management. FAO has had a close collaboration with the Convention on Biological Diversity (CBD) in facilitating implementation of CBD decisions related to the International Pollinator Initiative (also known as the International Initiative on the Conservation and Sustainable Use of Pollinators),[16] recognizing the need for more global coordination of pollinator-relevant policies (Drivdal and van der Sluijs, 2021). Under the Plan of Action (2018–2030) for pollinators, Element 1: Enabling policies and strategies, activity A1.1 (Safeguard and promote wild and managed pollinators into the broader policy agendas focused on sustainable development) explicitly mentions strengthening the link between human health, nutritious diets and pesticide exposure and reducing perverse incentives relating to pesticide use. Activity A1.2 (Implement effective pesticide regulation) is entirely dedicated to improving regulation of pesticide use. Under Element 2: field-level implementation, activity A2.1 (Co-design (with farmers, beekeepers, and land managers) and implement pollinator-friendly practices in farms and grasslands) discusses implementing pollinator-friendly practices around pesticide use. Under Element 4: Monitoring, Research and Assessment, activity A4.2 (Research) mentioned the gaps in knowledge we have around pesticides and potential cumulative/synergistic effects of pesticides with other chemicals and/or pressures of pollinator loss.

6.3. Kunming-Montreal Global Biodiversity Framework

The Kunming-Montreal Global Biodiversity Framework, adopted in December 2022, sets out four global overarching goals and twenty-three action-oriented targets involving nature and ecosystem service conservation to 2030; 196 Parties to the CBD adopted this framework. Of the 23 targets, Target 7 on pollution is most relevant to addressing the issues regarding pesticide use. Target 7 states "Reduce pollution risks and the negative impact of pollution from all sources, by 2030, to levels that are not harmful to biodiversity and ecosystem functions and services, considering cumulative effects, including: reducing excess nutrients lost to the environment by at least half including through more efficient nutrient cycling and use; reducing the overall risk from pesticides and highly hazardous chemicals[17] by at least half including through integrated pest management, based on science, taking into account food security and livelihoods; and also preventing, reducing, and working towards eliminating plastic pollution." The final version of the adopted text for this target changed from the more stringent ambitious goal for pesticides from "Reduce pollution from all sources to levels that are not harmful to biodiversity and ecosystem functions and human health … and pesticides by at least two thirds" in the first draft of the post-2020 global biodiversity framework[18] to "reducing the overall risk….by at least half" appearing in the approved version. The challenge for countries is now to implement a monitoring framework that can effectively monitor the use of pesticides and "risk" using appropriate and specific, measurable, achievable, relevant, and time-bound (SMART) indicators. An expert meeting on developing a headline indicator 7.2 (under Target 7 on pollution) recommended Aggregated "Total Applied Toxicity (TAT) is a suitable indicator for reducing the effects of pesticides on key biota[19]", the deliberations on this headline indicator will be concluded at the CBD's sixteenth Conference of the Parties (October 2024).

7. Future research and policy needs

Although there has been a concerted effort to increase the basic information and accurate reporting of pesticides and impacts on pollinators in regions outside of North America and Europe – many large gaps still exist despite several major studies from some regions in Asia and South America since 2016. Research on insecticides other than neonicotinoids has also gained some momentum and several studies have looked into the impacts of a number of alternative "environmentally safer" pesticides derived from plant products. More research is needed, however, on a number of pyrethroid pesticides (O'Reilly and Stanley, 2023), sulfoximine-based insecticides (Tamburini et al., 2021b), emerging active ingredients; biopesticides (Cappa et al., 2022) and nanotechnology-based pesticides (Muneer et al., 2023). Research is also needed regarding the synergistic impact of co-exposure to mixtures of pesticides (insecticides, fungicides, herbicides, co-formulants) on pollinators. Similarly, we need to understand more about how the interplay among multiple biotic stressors (bee forage and nutrition deficits, pathogens, and parasites) and pesticides impact pollinators across life-history stages, castes, or sexes. Studies that integrate assessment of the impacts of pesticide exposure in laboratory, semi-field experiments and under variable field or landscape contexts would be useful to provide a more systemic assessment of risk from individual organism to population scales. The lack of longer-term studies of the cumulative effects of pesticide exposure on a range of pollinators (and other non-target organisms) make it impossible to forecast the temporal risk of pesticide use for biodiversity and related ecosystem functions. There remains a clear need to understand the ecosystem level consequences of impacts of pesticides on pollinators and other non-target organisms. These might include study of knock-on effects via ecological interaction networks for wild plant reproduction and community dynamics or spillover impacts of pesticide use from point (e.g., glasshouse horticulture) or diffuse sources (e.g., open crop fields) on surrounding soil and water systems. These knowledge gaps collectively represent a problem for the achievement of sustainability goals and targets of national and international policies and multilateral environmental agreements (e.g., CBD).

CRediT authorship contribution statement

P. Basu: Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **H.T. Ngo:** Writing – review & editing, Writing – original draft, Conceptualization. **M.A. Aizen:** Writing – review & editing. **L.A. Garibaldi:** Writing – review & editing, Methodology. **B. Gemmill-Herren:** Writing – review & editing. **V. Imperatriz-Fonseca:** Writing – review & editing. **A.M. Klein:** Writing – review & editing. **S.G. Potts:** Writing – review & editing, Conceptualization. **C.L. Seymour:** Writing – review & editing. **A.J. Vanbergen:** Writing – review & editing.

Declaration of competing interest

There are no financial and personal relationships with other people or organizations that could inappropriately influence (bias) this work and the authors have nothing to declare.

Data availability

The dataset to run statistical analyses can be found here:

[16] CBD COP Decisions: V/5 and XIII/15 paragraph 1

[17] Retrieved April 2024 from https://www.cbd.int/gbf/targets/7/ - the definition of "highly hazardous chemicals" agreed by CBD parties is – "they are chemicals that pose a significant acute or chronic risk to the environment or people."

[18] Retrieved January 2023 from https://www.cbd.int/doc/c/abb5/591f/2e46096d3f0330b08ce87a45/wg2020-03-03-en.pdf

[19] CBD/SBSTTA/26/INF/18 and CBD/SBSTTA/26/2

https://zenodo.org/records/13836762.

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Appendix A. Supplementary data

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