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Review

Beyond the field: How pesticide drift endangers biodiversity☆

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A B S T R A C T

Airborne pesticide drift poses a substantial environmental threat in agriculture, affecting ecosystems far from the application sites. This process, in which up to 25% of applied pesticides are carried by air currents, can transport chemicals over hundreds or even thousands of kilometers. Drift rates peak during the summer months, reaching as high as 60%, and are influenced by various factors, including wind speed, temperature, humidity, and soil type. Pesticide volatilization is a significant concern, occurring 25 times more frequently than surface runoff. Under certain conditions, it can result in chemical losses of compounds like metolachlor and atrazine that are up to 150 times higher. These drifting pesticides have profound impacts on biodiversity, harming non-target plants, insects, fungi, and other organisms both near application sites and in distant ecosystems. Pesticide drift has been linked to over 50% reductions in wild plant diversity within 500 m of fields, reducing floral resources for pollinators. Despite growing evidence of these effects, the long-term consequences of airborne pesticides on biodiversity remain poorly understood, especially in complex field conditions with multiple pesticide applications. Addressing this requires urgent measures, such as improved meteorological tracking during applications, adoption of biopesticides, and integrated pest management strategies. This review highlights the pressing need for research to quantify airborne pesticides' ecological impacts, advocating for sustainable practices to mitigate environmental damage.

1. Introduction

Pesticides—bioactive compounds aimed at controlling pests and safeguarding crops—have been integral to agriculture for centuries. While the first documented use of inorganic fungicides dates back to 1637, the widespread adoption of synthetic pesticides began in the mid-20th century (Strassemeyer et al., 2017). The largest introduction of commercially available pesticides took place between the 1970s to the 1990s leading to a 50% increase in their use as compared to the previous decades (Andreazza et al., 2015; Kim et al., 2023). Data from the Food and Agriculture Organization (FAO) shows that, on a global scale, pesticide sales have nearly doubled in 2021 (3.5 million metric tons) compared to 1990 (1.7 million metric tons), despite declines reported in some individual countries, with large global variations in the distribution of pesticide use between countries. According to 2021 statistics (Statista, 2023), the Americas consume more than half of the world's

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agricultural pesticides, followed by Asia (27.7%) and Europe (14.3%), while Africa and Oceania combined use less than 8% of the global sales of pesticides.

The complexity of pesticide use is evident in the huge number of registered formulations. The US Environmental Protection Agency (EPA) currently lists around 600 active pesticide ingredients, with over 1000 different pesticide formulations in use worldwide (US EPA, 2013; WHO, 2022). The overuse of synthetic pesticides has unintentionally contaminated unintended areas, harmed non-target species and disrupted ecosystems that support agricultural productivity. These undesirable effects often occur through a phenomenon called pesticide drift. During pesticide spraying, some particles are released into the air as a result of spray drifting or evaporation from plants or the soil surface. These airborne pesticides are then carried by wind beyond the intended area, causing them to reach non-target areas. Consequently, this drift exposes a wide range of non-target living organisms such as beneficial insects, birds, and other animals that live near to/or far from the incurred areas (Saroop and Tamchos, 2024). When non-target species are harmed, the intricate balance of the ecosystem becomes, consequently, compromised. These ecosystems, which often contain natural predators and pollinators essential to crop health, become unstable, reducing their ability to support sustainable agriculture.

However, despite existing concerns, information on the full impact of pesticides on non-target species remains limited. This lack of knowledge hinders effective risk management. Current environmental risk assessment (ERA) schemes often overlook the impact on ecosystem functions and biodiversity, especially in remote areas where pesticide drift is the main exposure route (Brühl and Zaller, 2019; Mauser et al., 2024). Likewise, the EU Directive No. 91/414/EEC excludes effects on non-target plants from herbicide risk assessments.

Compounding the problem are inadequate preventive actions, and inconsistent remediation efforts (ANSES, 2020; Langenbach and Caldas, 2018). The lack of comprehensive monitoring makes it difficult to understand the true scope of the impacts of pesticide drift on unintended targets. Only by closing these knowledge gaps and implementing robust monitoring systems, can we develop solutions that minimize the harmful effects and ensure a sustainable future for both agriculture and natural ecosystems. The broader environmental impact must be carefully managed to preserve the ecological equilibrium on which agriculture ultimately depends (Khan et al., 2023). Efforts to reduce these impacts should primarily target pesticide contamination in water and air, as runoff and drift are the main pathways for pesticide movement offsite (Chagnon et al., 2015; Gunstone et al., 2021).

This review provides an understanding of the consequences of airborne pesticide drift on biodiversity, particularly in vulnerable areas. It addresses the interconnectedness of these impacts and illustrates the complex threats that pesticide drift poses to biodiversity across multiple ecosystems. It synthesizes existing data from various ecological contexts to provide a thorough analysis of how pesticide drift impacts diverse biological systems.

2. Pesticide drift & deposition

Pesticide drift occurs due to evaporation or improper spraying followed by wind carrying of the drifted particles. The drifted particles can travel through the air over distances of hundreds of kilometers, and in some cases, exceed a thousand kilometers (Kassianov et al., 2017; Mayer et al., 2024), Fig. 1.

It is estimated that a quarter of the pesticide sprayed end up being drifted (Aktar et al., 2009). More than 100 airborne pesticides were detected in a biosphere reserve in Germany, of which 28 were not approved for use in the country, captured pendimethalin concentrations ranging up to 18.3 ng/m3 (Kruse-Plaß et al., 2021). In northeastern Italy, aerosol samples collected from Roncade and the Col Margherita Observatory revealed 14 distinct pesticides. Concentrations ranged from cyanuric acid, primarily detected at Roncade with a mean concentration of 10 ± 10 ng/m^3, to glyphosate and fosetyl-aluminium at 0.1 ± 0.2 ng/m^3 and 0.1 ± 0.1 ng/m^3, respectively (Feltracco et al., 2022a,b). Moreover, A study by Zaller et al. (2022) conducted in eastern Austria detected 67 active pesticide components in air samples across surveyed sites, with an average of 10–53 components per site. Chlorpyrifos-ethyl was the most prevalent insecticide, appearing in 93% of the samples. The highest concentrations measured included the herbicide prosulfocarb (725 ± 1218 ng per sample), the fungicide folpet (412 ± 465 ng per sample), and the insecticide chlorpyrifos-ethyl (110 ± 98 ng per sample). Table 1 shows that glyphosate concentrations are highly irregular in different regions, from very high concentrations in Malaysia at as high as 42,960 ng/m^3, to very low concentrations in Europe and North America, usually below 10 ng/m^3. High concentrations detected in Malaysia may well indicate heavy agricultural use, perhaps in combination with less restrictive pesticide laws. The low but consistent detection of organochlorine insecticides from different regions suggests the possible role of pesticide drift from areas where these substances are still used to areas where widespread bans are observed. In addition, these levels are indicative of the sustaining presence of pesticides in the natural environment, especially within countries that are highly industrialized where these chemicals may have been used much more heavily in the past. Table 1 provides data on pesticides detected in air samples from several parts of the globe.

Detectable levels of pesticides in pristine regions, such as remote parts of Brazil, illustrate how atmospheric transport can disseminate the chemicals long distances from their original site of application and therefore contaminate regions with little or no direct use of pesticides. The diverse pesticide families detected included herbicides, insecticides, and fungicides. This diversity underlines the global scale of pesticide application, and the complex mixture of chemicals present in the atmosphere due to varied regional agricultural needs and practices.

Pesticides in air samples have been traced back to agricultural

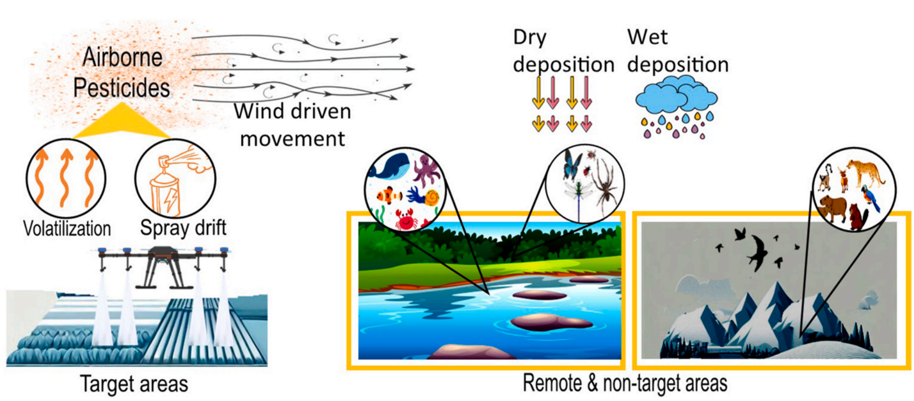

Fig. 1. Sources of airborne pesticides and methods of deposition.

Table 1
Reported data on pesticides detected in air samples from several parts of the globe.

Type of Pesticide	Concentration	Region/ Location	Reference
Glyphosate	0.1–0.3 ng m^{-3}	North-Eastern, Italy	(Feltracco et al., 2022a,b)
Cyanuric acid (metabolite of Cyromazine insecticide)	10.0–20.0 ng m^{-3}		
Fosetyl-aluminium (fungicide)	0.1–0.2 ng m^{-3}		
Glyphosate	42960 ng m^{-3}	Malaysia	Morshed et al. (2011)
Glyphosate	<0.01–9.1 ng m^{-3}	Mississippi River Basin, US	Chang et al. (2011)
Glyphosate	0.18–1.04 ng m^{-3}	Alpes-Cote-d'Azur, France	Ravier et al. (2019)
Different Pesticide families	1.15–30.51 ng m^{-3}	Pas-de-Calais, France	Prouvost and Declercq (2005)
Organochlorine Pesticides	3.5×10^{-7}–1.283×10^{-3} ng m^{-3}	Toronto, Canada	Motelay-Massei et al. (2005)
Acetochlor (Acetanilide Herbicides)	1.84 ng m^{-3}	Centre Region, France	Coscollà et al. (2010)
Pendimethalin (Dinitroaniline Herbicides)	1.93 ng m^{-3}		
Trifluralin (Dinitroaniline Herbicides)	1.32 ng m^{-3}		
Chlorothalonil (Chloronitrile fungicides)	12.15 ng m^{-3}		
Organochlorine insecticides	4.5×10^{-6}–3.00×10^{-4} ng m^{-3}	Toronto, Canada	Farrar et al. (2005)
Organochlorine insecticides	0.5×10^{-6} – 1.01×10^{-4} ng m^{-3}	North–South Transect, Chile	Pozo et al. (2004)
Organochlorine insecticides	7.0×10^{-8}–4.64×10^{-4} ng m^{-3}	North America	Shen et al. (2005)
Organochlorine insecticides	6.0×10^{-7}– 6.6×10^{-5} ng m^{-3}	North, Canada	Wania et al. (2003)
Organochlorine insecticides	1.7×10^{-7} – 8.17×10^{-4} ngm^{-3}	Toronto, Canada	Motelay-Massei et al. (2005)
Organochlorine insecticides	0–1.775×10^{-3} ng m^{-3}	Across Europe	Farrar et al. (2006)
Organochlorine insecticides	1.3×10^{-6} – 7.30×10^{-4} ng m^{-3}	Neratovice, Czech Republic	Klánová et al. (2006)
Organochlorine insecticides	18.4–7469 ng m^{-3}	Bitterfeld, Germany	Wennrich et al. (2002)
Organochlorine insecticides	2.60×10^{-4} – 6.300×10^{-3} ng m^{-3}	North Pacific Ocean	Prest et al. (1995)
Fenpropidin Fungicides	0.07–0.50 ng m^{-3}	Villiers-en-Bois, France	Décuq et al. (2022)
Epoxiconazole (Triazole Fungicide)	<DL– 0.27 ng m^{-3}		
Metrafenone (Benzophenone Fungicide)	<DL– 0.09 ng m^{-3}		
S-metolachlor (Chloroacetamide Herbicide)	031–1.08 ng m^{-3}		
Pendimethalin (Dinitroaniline Herbicide)	1.21–22.44 ng m^{-3}		
Tau-fluvalinate (Pyrethroid Insecticide)	<DL– 0.17 ng m^{-3}		
Organochlorine insecticides	1.44×10^{-5} – 1.52×10^{-4} ng m^{-3}	Shenzhen, Harbin, and Mohe, China	Wang et al. (2023)
Different Pesticide families	0.05–25.4 ng m^{-3}	Northwestern Mississippi, USA	Majewski et al. (2014)
Organochlorine insecticides	< LOQ – 3.52×10^{-4} ng m^{-3}	Central Europe	Shahpoury et al. (2015)
Organochlorine insecticides	7.2×10^{-6} – 8.29×10^{-5} ng m^{-3}	A coastal area, Turkey	Cindoruk and Ozturk (2016)
Organochlorine insecticides	180–2950 ng m^{-3}	Kocaeli, Turkey	Binici et al. (2014)
Endosulfan	1.27×10^{-3} – 3.2×10^{-3} ng m^{-3}	Pristine areas (Brazil)	Guida et al. (2018)
Cypermethrin	1.48×10^{-4} – 8.81×10^{-4} ng m^{-3}		
Chlorpyrifos	6.7×10^{-5} – 2.70×10^{-4} ng m^{-3}		
Triazine Herbicides	4.0×10^{-6} – 2.100×10^{-3} ng m^{-3}	Central Chile, Chile	Cortes et al. (2020)
Organophosphate Insecticides	1.05×10^{-6} 14.624×10^{-3} ng m^{-3}		
Metolachlor (chloroacetanilide herbicide)	13.0×10^{-6} – 4.03×10^{-4} ng m^{-3}		
Pendimethalin (Dinitroaniline Herbicide)	2.65×10^{-4} – 14.927×10^{-3} ng m^{-3}		
Tebuconazole (Triazole Fungicide)	1.2×10^{-5} – 6.00×10^{-4} ng m^{-3}		

practices as the main source. For example, it was found that concentrations of pesticide drift in air samples increased with the area of arable land (Hallmann et al., 2017; Zaller et al., 2022). However, several surveys showed that incidents of off-target pesticide movement are underreported (Bish et al., 2021).

Pesticide drift through volatilization is a significant concern, with studies indicating that it can be 25 times more prevalent than drift caused by surface runoff (Karlsson and Arvidsson, 2015). In an 8-year study, losses of metolachlor and atrazine through volatilization were 150-fold and 130-fold greater than losses through runoff under favorable conditions (Gish et al., 2011). The rate of volatilization is, however, governed by several factors including vapor pressure, and water

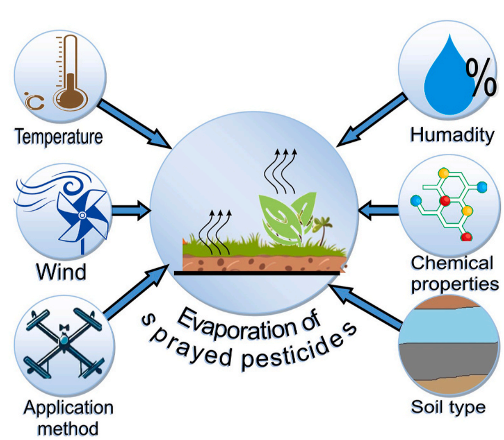

Fig. 2. Factors affecting the evaporation rate of sprayed pesticides.

solubility (Boonupara et al., 2023; Doan Ngoc et al., 2015). This is evident in the varying rates of volatilization observed for different pesticides, Fig. 2.

However, prediction based on physico-chemical properties is not always straightforward. For example, although the probability of drift is low based on its vapor pressure, dicamba (a selective systemic herbicide) has a high tendency to drift and has been involved in numerous mist drift incidents (Zaller and Brühl, 2021). This highlights the role of other factors such as meteorological conditions. For example, drift rates were higher in summer (up to 60%) compared to autumn (up to 18%) (Karlsson and Arvidsson, 2015). Moreover, Linhart et al. (2021) reported peak seasonal drift rates in spring and summer in South Tyrol, Italy, with 76% of detected residues being endocrine active. Residue levels varied significantly by season ($p < 0.001$): 25 residues at 83% of sites in spring, nine at 79% in summer, three at 50% in autumn, and four at 17% in winter. Wind speed and direction can significantly impact drift patterns (Desmarteau et al., 2020). It's also important to consider regional variations in humidity. For example, while higher air humidity increases the volatilization of pesticides from the soil surface, it improves the uptake of pesticides by plants (Farha et al., 2016). Additionally, Linhart et al. (2021) reported a 76% contamination rate in playgrounds in Venosta Valley, Italy, with the highest levels in Low Adige. Pesticide levels rose with proximity to apple orchards, rainfall, and wind speed, while higher irradiance, opposite wind direction, greater distance from agriculture, and restrictions on high-wind applications reduced contamination.

The type of soil can also affect the rate of volatilization. For example, only 2% loss of applied chlordane was lost via volatilization from dry soil after 2 days, whereas the volatilization of the same substance from sandy/wet soil was 50% of the applied amount (Taylor and Spencer, 1990). Experiments showed that pesticides are more likely to volatilize when the amount of water in the soil is above some threshold value; below this threshold, the vapor density of these pesticides drops off sharply, which serves to decrease volatilization (Schneider et al., 2013). At low moisture, pesticides are more strongly bound to the adsorption sites on soil particles, which reduces their fugacity and hence their vapor density. With an increase in soil moisture, the adsorption sites release the pesticides by which volatilization can again take place (Grover et al., 1985; Jansma and Linders, 1995). This process could explain the highest efficacy of pesticides following rainfall or irrigation, since soil moisture normally triggers the release of adsorbed chemicals into the environment where they are available for volatilization and plant uptake. Experimental studies showed that pesticide volatilization was maximum at 100% relative humidity even when water loss was zero underlining the idea that volatilization is independent of water evaporation (Voutsas, 2007). Increased temperatures generally increase the rate of volatilization, but this effect is complicated in dry soils since, once the soil surface dries and can no longer cool through the evaporating water, soil temperatures rise. This heating itself enhances adsorption of pesticides, thereby reducing volatilization, a phenomenon wherein an increase in temperature does not simply correlate with an increase in volatilization due to complex interactions with soil moisture levels and adsorption dynamics (Nash, 1983).

On the other hand, dry soils with very low moisture content can promote secondary drift, where pesticides bound to small, dry soil particles are transported by wind. In contrast to volatilization, secondary drift involves particulate matter carrying pesticide residues, which can be transported over longer distances (Franke et al., 2010). These conditions become especially pronounced under high temperatures and arid conditions where wind readily lifts pesticide-bound particulates, creating a risk of drift beyond the intended application area (Meftaul et al., 2020). By considering these factors, the differences in pesticide loss and transport mechanisms from various soil types become clearer. These observations point to the importance of understanding the complex interactions between different factors.

While the peak of pesticide volatilization occurs in the first 24 h after application, it may continue for several weeks (or sometimes longer), and it sometimes displays a diurnal cycle (Karlsson and Arvidsson, 2015; Prueger et al., 2005).

International regulatory bodies recommend pesticide application under stable meteorological conditions to reduce pesticide drift (IFAS, 2022). However, stable atmospheric conditions, in turn, can increase stability of pesticide particles in the atmosphere, facilitating their subsequent deposition in non-target areas (Grace and Tepper, 2021). This phenomenon is called secondary movement of pesticides (Bish et al., 2019). Therefore, the complexity of natural systems makes it difficult to isolate the effects of individual variables. Acknowledging these limitations allows for a more precise interpretation of the results and underscores the need for continued research to improve our understanding of the dynamics of pesticide drift. A recent study suggests that pesticide drift could cover an area of up to 10,300 km^2 (El Afandi et al., 2024).

Airborne pesticides can eventually return to the ground through two main processes: wet deposition by rain, snow, or fog, and dry deposition by gravity or turbulent diffusion. Wet deposition is a major contributor to deposition of air particles-bound chemicals (Đurašković et al., 2012; Pan et al., 2017). An example of wet deposition is acid rain. Wet deposition attracts water-soluble compounds and ions more efficiently than hydrophobic organics and gaseous pollutants (Pan et al., 2017). Its deposits are up to 80% of hydrophilic particles as compared to 20% of hydrophobic particles; in comparison, dry deposition deposits are roughly 1:1 (46% of hydrophilic and 54% hydrophobic particles) (Matsumoto et al., 2019). Table 2 shows that wet deposition contributes significantly to the deposition of airborne pesticides in neighboring non-target areas and remote locations.

For chemical contaminants with high vapor pressures, low molecular weights, and low adsorption coefficients, dry deposition is the main pathway for settlement (Zhang et al., 2012). Dry deposition has been shown to be much more effective than wet deposition in reducing surface level concentrations of both semi-volatile organic compounds and secondary organic aerosol (Knote et al., 2015). The rate of dry deposition is influenced by various atmospheric conditions. Wind direction plays a crucial role in dry deposition (Desmarteau et al., 2020). The rate of deposition fluxes during daytime and night has been shown to vary with studies reporting contradictory observations, possibly due to diurnal variability of meteorological factors (Zhu et al., 2016). The characteristics of the terrain can also affect the rates of dry deposition. For example, vegetable or rough surfaces can aid in the dry deposition of pesticides by providing a larger surface area with a higher frictional property (Elsaesser et al., 2011; Xing and Brimblecombe, 2019).

Airborne pesticides can exist as gaseous aerosols or be adsorbed on airborne particulate matter (PM), where PM may serve as a carrier in transporting and depositing pesticides (Gao et al., 2024; Nascimento et al., 2017). PM can also be classified into fine and coarse particulate matter based on their size (World Bank, 1999). Under typical conditions, PM's atmospheric lifetimes range from 3 to 10 days (Nascimento et al., 2017). The size of PM significantly influences its removal process as gravity primarily affects coarse particulate, while diffusion dominates for fine particulate (Farmer et al., 2021). This is reflected in how different PM sizes are removed from the air. For example, wet deposition, where rain or snow washes away particles, is more efficient for fine particulate, removing around 92%. Conversely, dry deposition, where PM particles settle due to gravity and air movement, is more effective for coarse particulate, removing around 63% (Wu et al., 2018). Temperature negatively correlates with velocity of dry deposition of PM (Wu et al., 2018). However, the tendency of semi-volatile pesticides to adsorb onto particulate matter, render them very resistant to oxidation extending their atmospheric lifetimes from few days to more than a month depending on the nature of the pesticide (Mattei et al., 2019; Pathak et al., 2022; Socorro et al., 2016). Staying longer in the air allows these substances to increase chemical stability and thus be transported to other geographic locations, including remote areas beyond their expected half-lives. Nevertheless, not all airborne pesticides settle on the

Table 2
Reported data on pesticides detected in rainwater samples from several parts of the globe.

Type of Pesticide	Concentration	Region/ Location	Reference
Different Pesticide families	0.011–0.10μgL^{-1}	Arizona, USA	Villagómez-Márquez et al. (2023)
Organochlorine insecticides	2.0–40 μgL−1	Anambra State, Nigeria	Okafor et al. (2019)
Organochlorine insecticides	0.001–3200 μgL−1	Kibaha district, Tanzania	Mahugija et al. (2015)
Organochlorines	0.041–7.06 μgL −1	India, Hisar	Kumari et al. (2007)
Different Fungicides	0.0014–0.05 μgL−1	Villiers-en-Bois, France	Décuq et al. (2022)
Different Herbicides	0.0005–0.17 μgL−1		
Pyrethroid Insecticides	0.008–0.05 μgL−1		
Glyphosate	<0.1–2.5 μg/L	Mississippi River basin, USA	Chang et al. (2011)
Synthetic pyrethroids	0.1–1.0 μgL −1	Hisar, India	Kumari et al. (2007)
Organophosphates	0.050–4.00 μgL −1		
Different Pesticide families	0.010–5.59 μgL−1	Alsace, France	Scheyer et al. (2006)
Glyphosate	<LOD – 8.2 μg L^{-1}	Argentina	Lupi et al. (2019)
Different Pesticide families	<LOD – 7.09 μgL −1	South Georgia, USA	Glinski et al. (2018)
Organochlorine insecticides	0.004–0.15 μgL^{-1}	Beijing, China	Yang et al. (2012)
Different Pesticide families	0.0001–0.43 μgL^{-1}	Singapore	Zhang et al. (2018)
Glyphosate	1.24–67.3 μgL^{-1}	Pampas region, Argentina	Alonso et al. (2018)
Atrazine	0.22–26.9 μgL^{-1}		
Triazine Herbicides	0.007–0.90 μgL^{-1}	Grüze, Switzerland	Bucheli et al. (1998)
Acetamide herbicides	0.012–0.19 μgL^{-1}		
Phenoxy acid herbicides	0.011–0.11 μgL^{-1}		
Triazine Herbicides	0.004–0.15μgL^{-1}	Pantanal Basin, Brazil	Laabs et al. (2002)
Organophosphate Insecticides	0.016–27.6 μgL^{-1}		
Pyrethroid insecticides	<LOD – 0.38 μgL^{-1}		
Organochlorine insecticides	0.004–0.33 μgL^{-1}		
Acetanilide Herbicides	0.006–0.87 μgL^{-1}		
Triazole Fungicides	0.017–0.18 μgL^{-1}		
Dinitroaniline Herbicides	0.003–0.32 μgL^{-1}		
Triazine Herbicides	0.036–1.03 μgL−1	Strasbourg, France	Scheyer et al. (2007)
Organochlorine Insecticides	<QL – 3.67 μgL−1		
Dicarboximide Fungicides	<QL – 5.59 μgL−1		
Acetanilide Herbicides	<QL – 0.12 μgL−		
Phenylurea Herbicides	<QL – 1.03 μgL−1		
Phenoxycarboxylic Acid Herbicides	<QL – 0.06 μgL−1		
Organochlorine pesticides	0.740–19. 97 μgL−1	Gdansk, Poland	Polkowska et al. (2009)
Organonitrogen and organophosphorus	0.020–5.20 μgL−1		

Table 2 (*continued*)

Type of Pesticide	Concentration	Region/ Location	Reference
Erbuthylazine (Triazine Herbicide)	0.003–0.52 μgL^{-1}	Lower Saxony, Germany	Hüskes and Levsen (1997)
Metolachlor (Chloroacetanilide Herbicide)	0.003–0.51 pg/L		
Metalaxyl (Anilide Fungicide)	0.006–0.48 μgL^{-1}		
Chlorothalonil (Chloronitrile Fungicide)	0.003–1.10 μgL^{-1}		
Atrazine (Triazine Herbicide)	0.023–0.21 μgL^{-1}	Province of Limburg, Netherlands	van Maanen et al. (2001)
Organochlorine Pesticides (β+γ-HCH, Endosulfan α & β, DDT)	0.003–0.14 μgL^{-1}		
Dichlorvos (Organophosphate Insecticide)	0.003–0.01 μgL^{-1}	Flanders, Belgium	Quaghebeur et al. (2004)
Diuron (Urea Herbicide)	0.001–1.47 μgL^{-1}		
Glyphosate (Phosphonate Herbicide)	0.003–0.48 μgL^{-1}		

ground, some remain afloat where they can cause direct poisoning through the skin or by inhalation which may disproportionately affect birds and flying insects (Authority (EFSA) et al., 2023; Katagi and Fujisawa, 2021).

3. Impact on biodiversity

3.1. Impacts on habitats and lower trophic levels

Biodiversity is very important in ecosystems because it promotes ecological stability and function. Healthy communities are made up of diverse species and groups that depend on each other and provide important services such as food and soil, which are essential for the continuation of life. Documented biodiversity loss due to human activities is alarming. Since 1970, a significant decline in biodiversity of 69% has been observed on average (Mahmoud and Gan, 2018; Ogidi and Akpan, 2022). The Living Planet Report 2020 has documented an average of 68% decline in the numbers of birds, amphibians, mammals, fish, and reptiles between 1970 and 2016 (Almond et al., 2020), as

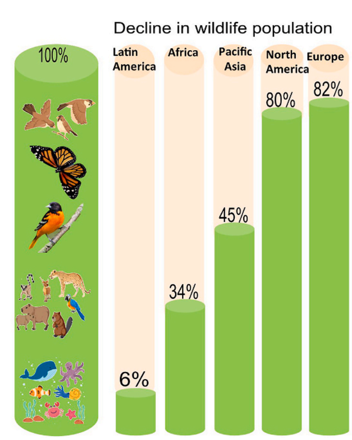

Fig. 3. Average decline in the relative abundance of monitored wildlife populations around the world between 1970 and 2018. (*A visualization made by the authors based on raw data published in the living plant report 2022*).

illustrated in Fig. 3.

For instance, amphibian species are particularly vulnerable to agrochemical exposure due to their permeable skin, which makes them susceptible to contaminants, leading to drastic declines in certain regions (Rajak et al., 2023). Aquatic habitats, such as streams and ponds near agricultural areas, have also seen significant declines in invertebrate diversity, including essential pollinators and insects that serve as key food sources for larger animals (Samways et al., 2020). The negative impacts of pesticide use on biodiversity and ecosystem services worldwide have been underestimated (Dudley et al., 2017). Evidence shows that pesticides are driving severe biodiversity declines, often acting in concert with additional stressors (Goulson et al., 2015). In addition, there are also important economic considerations concerning the negative impact of pesticides. For example, pesticides' effects on nontarget organisms, such as pollinators. While pesticide drift affects regions adjacent to agricultural fields, which are generally more contained in scope, long-range transport involves extensive movement, impacting ecosystems and species far from the application site. long-range transport with long half-lives extend the negative impact of pesticides on biodiversity in protected and remote areas (Martín-López et al., 2011).

Pesticide drift is adversely affecting natural habitats bordering agricultural fields as well as in remote ecosystems (de Jong et al., 2008; Schmitz et al., 2014). A three-year study investigating the effect of low-dose herbicide drift (atrazine and/or tripuron-methyl) on adjacent fallow fields found that these herbicides harm wild plants and reduce plant diversity overall (Qi et al., 2020). Although the lethal effects of drifted herbicides were only observed up to 6–500 m from the edge of the sprayed area, their impact was more widespread, with damage symptoms occurring at distances exceeding this range (Boutin et al., 2012; Zaller and Brühl, 2021). A series of field trials showed that the impact of herbicide drift on sensitive life stages of non-target vascular plants is estimated to have exceeded the 50% effect level on 59% of adjacent linear landscape elements such as ditch banks and hedgerows (de Jong et al., 2008). Herbicides, particularly glyphosate, dicamba, and 2,4-D sprays, have caused significant damage to many non-target plant species, often at sensitive life stages e.g. during seedling or flowering stages (Mohseni-Moghadam et al., 2016). Milkweed, a plant essential to monarch butterflies, often suffers glyphosate drift from farm areas. The resulting decline in milkweed plants has been one of the contributing factors to declining monarch populations since this plant helps in the growth and development of caterpillar stages of life during its life cycle (Pleasants and Oberhauser, 2013). Leafy vegetables like perilla leaves and Welsh onion had the highest residues of drifted pesticides, likely due to their large exposed surfaces (Kim et al., 2023). Vegetation and other valuable plant species support certain wild populations and provide essential food, cover, and breeding grounds for herbivorous insects, forming the base of a healthy food web (Barbercheck and Wallace, 2021; Kumar et al., 2021). Disruption of this balance is particularly worrying given that these systems harbor a significant portion of total plant biodiversity. Over the past century, Europe has witnessed a decline in weed flora and overall biodiversity due to extensive herbicide use in agriculture, particularly on arable crops. Studies reveal a significant decline in weed seedbank diversity with some species having shown a 50% reduction in seedbank size (Andreasen et al., 2018; Marshall et al., 2003).

An investigation into the effect of low-dose glyphosate spray drift on flowering in non-target plants within an agricultural setting showed that glyphosate significantly reduced the total number of flowers produced by two wild plant species (red clover and trilobite) susceptible to drift (Strandberg et al., 2021). The effect on flowering, however, varied between plant species. In addition, herbicides can cause short-term declines in plant species diversity by altering vegetative structure, and potentially changing plant successional trajectories (Guynn Jr et al., 2004; US EPA, 2015). Although in some cases, recovery from damage from simple or temporary exposure to drift pesticides has been reported (Buehring et al., 2007), repeated exposure can hamper this advantage. Moreover, not all species have the capacity to heal from minor wounds (Ramos et al., 2021).

3.2. *Impact on insects, birds and aquatic life*

Although, herbicide drift primarily impacts lower-order trophic levels, including primary producers such as plants and algae, it indirectly impacts higher trophic levels through changes in food availability and quality of habitat for herbivores, insectivores, and predators. By causing a reduction in weed cover, herbicides deprive insects of food plants, intensify insect pressure on crops and, as consequence, may endanger species at higher trophic levels such as birds, bats, and mammals who feed on insects. Long-term pesticide drift leads to the gradual, cumulative loss of insect diversity within the non-target areas. First, there will be a cascading effect through the trophic levels, as primary producers and pollinators decline, thereby reducing food sources and habitat quality for a wide range of other species. Some field studies have observed that, as plant diversity declines due to herbicide drift, so does the diversity of herbivores and predators and results in less balanced and more often pest-prone ecosystems. Thus, even though herbicides, insecticides, and fungicides target different organisms, their combined effects across multiple trophic levels can result in overall ecosystem destabilization (Sánchez-Bayo, 2011). A 27-year long study across 63 protected natural areas in Germany revealed an alarming 76% decrease in the biomass of flying insects, with peak summer declines reaching 82% (Hallmann et al., 2017). The combined impact of agricultural chemical pollution and other threats has led to a 40% decline in insect species across temperate regions within a few decades (Cardoso et al., 2020). Similar trends were observed for insect populations (including bugs, spiders, butterflies, and beetles) which declined by an alarming 50% between 1970 and 2004 in varying degrees (Goulet and Masner, 2017; Sánchez-Bayo, 2021). Spray drift is identified as a primary cause of non-target butterfly exposure to pesticides, as these chemicals drift from cultivated fields to regions with greater resources for butterflies, including hedgerows, wildflower patches, or even adjacent nature reserves (Zhong et al., 2010). Hedgerows and the spaces between agricultural lands are home to a large number of butterfly species (Krauss et al., 2003). Thus, any disruption in these habitats can affect butterflies that live in hedgerows (Kjær et al., 2014). Between 2000 and 2009, the number of common butterflies on farmland under observation decreased by 58% (Brereton et al., 2011). Larvae of P. brassicae and P. napi were subjected to the pesticide diflubenzuron in a simulated laboratory experiment at concentrations that were identical to realistic concentrations resulting from field spray drift (Davis et al., 1991). The experiments showed that the larvae were highly sensitive to the pesticide diflubenzuron, making them a suitable indicator for studying the effects of pesticide drift on butterflies in general.

Since insects are a vital food source for many birds, this decline can be linked to the reported 50% reduction in the numbers of insect-feeding birds of the agricultural lands at the European level (Brühl and Zaller, 2019). Farmland bird populations have been experiencing a dramatic decline for decades, and bird mortality was observed to significantly increase due to a lack of insects readily available (UN, 2024; Youth, 2019). In the European Union alone, the farmland bird index showed a 17% decline between 2000 and 2018, which translates to an estimated loss of 560–620 million birds since 1980 (Burns et al., 2021; Eurostat, 2020). After 1990, the use of insecticides and herbicides increased in Saxony-Anhalt, Germany. This was coupled with a reduction in the area covered by grassland, resulting in a decline in biodiversity. For instance, populations of red kite (Milvus milvus) have decreased from 40 nesting pairs to about 20 pairs per 100 km2 (Nicolai et al., 2017). Similar trends have been observed in the United States, where the deaths of millions of birds are likely linked to pesticides (Ali et al., 2021). However, the full extent of bird mortality is likely much higher due to undetected deaths. Furthermore, impact of pesticide drift on specific bird species like bald eagles, hawks, and owls is particularly concerning (Rich et al., 2020; Terziev and Petkova-Georgieva, 2019). Although pesticides concentrations in air may, in many occasions, be below lethal levels, sublethal exposure to pesticides can cause long-term health problems in birds, including weight loss, increased susceptibility to disease, decreased reproductivity and nest abandonment (Gibbons et al., 2015; Jahn et al., 2014). Other important effects that are not easy to notice include behavioral changes and endocrine disorders (Köhler and Triebskorn, 2013a; Tavalieri et al., 2020). In an interesting study, organic pesticides were discovered in the feathers of three species of migratory birds (Xiao et al., 2023). The study found that birds with longer migrations tended to have higher levels of pesticides. These results demonstrate the potential for increased exposure of migratory birds to pesticides through the air. It has been estimated that pesticides and heavy metals in the air and surface waters are a major threat to more than 1211 bird species that are recognized as endangered with extinction worldwide (Isenring, 2010a).

Over one-third of crop production globally relies on animal pollination (IPBES, 2016), and studies from Britain and the Netherlands indicate declines in bee populations correlated with increased insecticide use (Biesmeijer et al., 2006; Rollin et al., 2016). Insecticides can decrease floral visitation rates, impact crop yields and reduce pollinator resilience. Additionally, the loss of pollinator species diversity—especially under continuous insecticide exposure—can destabilize pollination services, ultimately affecting food security (Biesmeijer et al., 2006). Herbicide drift may not affect pollen quality, but it can delay flowering and reduce the number of flowers, ultimately limiting resources available for pollinators (Brühl and Zaller, 2021; Carpenter et al., 2020).

Drifted pesticides can also settle on bodies of water, harming aquatic life and potentially disrupting entire aquatic food webs (Mishra et al., 2023; The European Environment Agency, 2024). Pesticide spraying was linked to residues ending up in surface water, with spray drift being a major culprit (Loewy et al., 2011). Pacific salmon and their aquatic habitats are in danger due to pesticide contamination of these areas through spray drift and surface runoff (Macneale et al., 2010). A California study revealed that pesticide drift has polluted more than 10% of streams in several regions (e.g. North Central Coast; (Rich et al., 2020). These waterways are now considered 'impaired' as their pesticide levels surpass established water quality standards, with neurotoxic insecticides particularly affecting stream-dwelling insects (State Water Resources Control Board, 2019). Spray drift from hexazinone, a triazine herbicide, was considered a potential threat causing a negative impact to the endangered California red-legged frog and its habitat (Isenring, 2010b). Amphibians can be exposed to pesticides through various ways, including drift events (Adams et al., 2021; Wagner et al., 2014). Their permeable skin makes them especially sensitive (Brühl et al., 2013). Although amphibians are not yet classified as endangered internationally or nationally, worrying population declines are already evident at the local level (Kyek et al., 2017; Petrovan and Schmidt, 2016). It has been estimated that pollution of surface waters with pesticides is a major threat to around 2000 amphibian species worldwide (Rowley et al., 2009).

3.3. Impact on wildlife

Pesticide drift allows for direct poisoning of wildlife through inhalation and skin contact. Secondary poisoning occurs when a predator consumes prey that has been contaminated or poisoned with pesticides web (Peillex and Pelletier, 2020; Syromyatnikov et al., 2020). Wildlife poisoning is most likely to occur from highly persistent pesticides that are resistant to weathering. Pesticides accumulate over time and make prey species more vulnerable to intoxication (Wobeser et al., 2004). The most common toxic effects studied include immune toxicity, endocrine disruption, reproductive effects, and behavioral changes. Organophosphate pesticides, for example, decrease egg production in wild birds and alter their feeding habits (Peris et al., 2023). However, only cases of poisoning can be followed up using existing wildlife disease surveillance systems. Consequently, chronic poisoning in wildlife has rarely been documented and primarily linked to organochlorines due to their persistence and stability in the natural environment (Kenntner et al., 2003). Although intoxication cases are frequently reported in the EU and US, there are only a handful of reports from Asia, Africa, and Central and South America although heavy use of pesticides and good biodiversity is present in these regions too. In non-avian species, endocrine disruption has been observed in river otters, bears, seals, sea lions, and beluga whales in organochlorine-contaminated areas (Köhler and Triebskorn, 2013b). However, the effects of DDT were difficult to separate from effects of other, non-pesticide organochlorines found in those environments (Bernanke and Köhler, 2009).

It should be noted here that the specific consequences for wildlife vary depending on the species and how they react to habitat modifications caused by drifting pesticides. Fig. 4 illustrates various processes by which drift pesticides can affect wildlife.

3.4. Impacts on biological communities and ecosystem services

Microorganisms are essential to ecosystem functions, contributing to nutrient cycling, decomposition, and overall soil health, thereby sustaining biogeochemical cycles (Sharma et al., 2023). However, pesticide exposure, particularly through spray drift and run off, can disrupt microbial communities and may have far-reaching effects on ecosystems. Spray drift is a primary pathway through which environmental microbes encounter pesticides (Stanley and Preetha, 2016). The effects of pesticides on microbial communities and their diversity include biomass reduction, growth disturbance, shifts in microbial community structure, and respiratory problems (Muturi et al., 2017). While the direct effects of pesticides on individual microbes are critical, understanding the cascade effects in complex ecosystems is even more important. Studies suggest that the direct effect observed in a simplified environment may not translate to real-world situations with diverse microbial communities (Jousset, 2012; Staley et al., 2015).

Fournier et al. (2020) studied further the effects of pesticides on essential soil microorganisms. Notably, the study found that the biopesticide influenced keystone taxa structuring the microbial network, while the synthetic pesticide altered interactions, favoring less efficient degraders of organic matter. These findings underscore the significant impact pest management products can have on soil microbiome function and demonstrate the value of high-throughput sequencing in assessing pesticide effects on non-target soil organisms.

Fungicides, aimed at controlling fungal pathogens, can inadvertently affect beneficial fungi and other microorganisms essential for nutrient cycling. For example, fungicide drift can disrupt soil fungi involved in mycorrhizal associations, impairing plant nutrient uptake and potentially reducing plant resilience (Ogidi and Akpan, 2023). Repeated fungicide exposure may also shift microbial community structures in the soil, leading to dominance by resistant strains, which could affect ecosystem stability and productivity (Ma et al., 2023).

4. The search for alternative pesticides with less impact on biodiversity

The environmental impact of current pesticides highlights the urgent need for less harmful alternatives. However, finding safer substitutes is challenging. For example, initial efforts to replace organophosphates and carbamates led to the use of neonicotinoid pesticides, initially deemed safer. While the shift from organophosphates to neonicotinoids reduced the toxic load for birds by 80.5% between 1990 and 2016, it simultaneously increased the toxic load for bees sixfold over the same period (Matsuda et al., 2020; Montaigu and Goulson, 2020). The recent move to dicamba, once described as an "environmentally friendly herbicide with little or no toxicity to wildlife and humans" has attracted

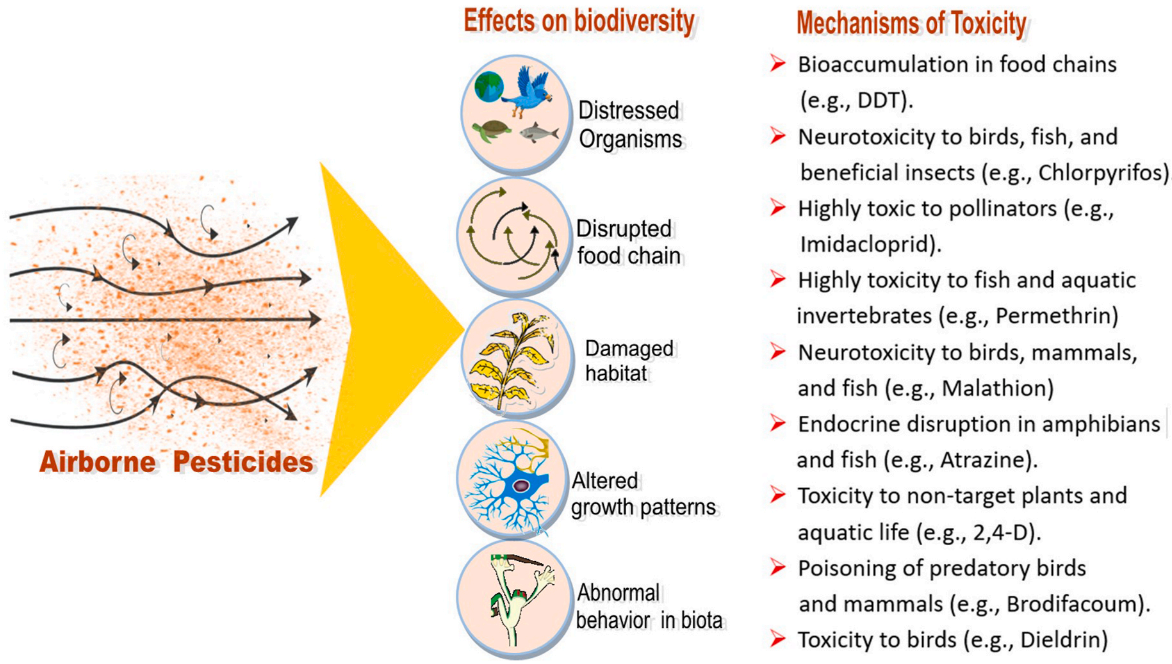

Fig. 4. Processes by which drift pesticides can impact biodiversity.

controversy due to its tendency to drift from treated fields into neighboring fields, causing damage (Johnson et al., 2023; US EPA, 2023). Sensitive vegetation and crops have been known to suffer significant harm from dicamba, 2,4-Dichlorophenoxyacetic acid, and glyphosate spray drift, particularly when best practices are not followed (Egan et al., 2014; Jones et al., 2019).

To address the need for safer alternatives, it must be emphasized here that current efforts must prioritize the development of agricultural practices that combine high productivity with environmental sustainability for the benefit of our planet (Schaeffer et al., 2018). In this context, green chemistry has the potential to contribute to efforts to find a sustainable approach to pest control while protecting biodiversity. However, there is still a gap between green chemistry and ecotoxicology that needs to be bridged to achieve a sustainable bioeconomy (Johann et al., 2022). The concept of “green toxicology” which refers to the application of predictive toxicology in the sustainable development and production of new less harmful materials and chemicals, can help bridging the gap between green chemistry and ecotoxicology (Crawford et al., 2017).

Biopesticides hold a promise as a sustainable alternative to synthetic pesticides as they can ensure high productivity without harming the environment (Fenibo et al., 2021). They fall into three main categories: microbial (like Bacillus thuringiensis), biochemicals (like essential oils), and plant-incorporated protectants (such as genetically modified plants). However, biopesticides still suffer from several limitations such as high cost, slower rate of action, inconsistent production standards, and vulnerability to environmental factors, that need to be controlled (Chandler et al., 2011; Essiedu et al., 2020). In the meantime, farmers can use crude plant extracts as a temporary solution (Stevenson et al., 2017).

In exploring alternatives to synthetic pesticides, it is essential to expand beyond genetically modified organisms and biochemicals to include methods rooted in agroecology and biological pest control. Agroecology advocates holistic, ecologically integrated pest management practices that reduce dependence on synthetic chemicals, promoting a resilient agricultural system. Biological pest control, through the use of natural predators, parasitoids, and pathogens, can provide effective pest management while supporting ecosystem health (Fenibo et al., 2021). Additionally, education and access to information are crucial for empowering farmers to adopt these methods effectively. Access to resources and guidance can help farmers transition to more sustainable practices, potentially reducing reliance on synthetic pesticides.

Pesticide regulation is based, to a great extent, on the paradigm of "presumed safe until proven hazardous" (Vogel, 2012). That is, pesticides are generally approved for marketing based on available determinations of safety and only subsequently restricted or removed from the marketplace when evidence develops to show that they present significant risks. The EPA, for instance, is mostly very reluctant to ban or put any limits on pesticides until substantial evidence of the damage caused can be presented (Jensen, 2015).

New research has shown that even when new evidence showed a risk in its use, including crop damage and health effects, the EPA has allowed the continued use of pesticides known for their environmental and health hazards, such as dicamba (Center for Food Safety, 2023). Besides, although many agricultural countries have managed to place more stringent bans or restrictions on toxic pesticides to protect environmental health in advance, the current policy seems not sufficient. Regulations must be able to better protect ecosystems and human health by requiring consideration of long-term effects before the wide acceptance of pesticides.

It is essential that pesticide manufacturers conduct extensive studies on the effects of new synthetic pesticides on ecosystems to prove that the pesticides will not cause harm before they are allowed to be used in the environment. Current regulations treat pesticides as harmless until proven to be harmful to environmental health. However, understanding the long-term environmental effects of pesticides before they are released into the environment must be considered.

Nevertheless, abandoning traditional pesticides is not easy: First, the regulatory framework governing pesticide approval generally favors established chemical solutions rather than innovative alternatives. Newer products can take an unusually long time to be approved before they reach the market. Second, farmers are often slow to adopt alternatives because of doubts about their effectiveness compared to traditional pesticides. Third, farmers may continue to use traditional pesticides because of their low direct costs and proven effectiveness, despite the environmental costs associated with their long-term use.

One aid that can accelerate the search for alternative and saver pesticides is the use of machine learning algorithms (ML). ML helps predict the toxicity of pesticides without their actual field application (Akhter and Sofi, 2022). This is a big step in the right direction for solving such intricate problems related to agricultural pesticide use. That makes it now a potent ally that aids stakeholders, policymakers, and academics in making proper decisions. ML provides the statistical inferences and methods which can be used for the detection and prediction of pesticide behavior in agricultural areas. Model-based regression techniques such as SVM, k-NN, ANN, CNN, LDA, DQA, and RF are recommended for toxicity prediction (Banaee et al., 2023). In mathematical ways, learning and optimizing moderating factors from only input and output data, machine learning in future will provide medium links (Anandhi and Iyapparaja, 2024).

Finally, there is the need to understand farmers' preference and handling habits of chemicals for the promotion of chemical pest management methods that do not harm the ecosystem and the health of farmers. In a study, only 24% of farmers expressed their interest in using lower risk pesticides, while about three quarters 76% of them expressed interest in using all kinds of available pesticides to control the most prevalent pests of their crops (Damalas et al., 2024). The application of lower-risk pesticides was found to be more common among better-educated farmers and those who apply pesticides more frequently (Damalas and Koutroubas, 2017).

5. Protecting biodiversity: strategies to minimize pesticides drift

Despite mitigation efforts, a European study revealed that over 73% of tested locations were contaminated with pesticides (Cech et al., 2023). Moreover, while regulations such as OECD guidelines ensure rigorous testing of herbicide drift onto crops, they often neglect the impact on wild plants (OECD, 2006). Developed countries are making progress towards minimizing local pesticide risks. The EU aims for a 50% reduction by 2023, China aims for pesticide-free agriculture by 2050, and the US has seen a 40% decrease in pesticide use over the last three decades (Candel et al., 2023; Melchior, 2022). Even in countries with declining overall use (e.g., UK), highly hazardous pesticides like glyphosate, acetamiprid, and imimazalil remain common (Claydon, S., Pesticide Action Network UK, 2023; Stroda and Garthwaite, 2022). Ironically, pesticide use has increased by more than 50% globally since 1990 (FAO-UN, 2022), leaving countries targeting reductions at increased risk.

Effective reduction of pesticide drift requires a multi-pronged approach, including.

- **Application methods**: Several studies have shown that the spray method is most effective when the optimal drop size is used for the intended target, highlighting the role of nozzle type in spray coverage and drift (Brühl and Zaller, 2021; De Cauwer et al., 2023; Grella et al., 2017). Grella et al. observed significant increases in downwind particle mass and number when using both Conventional (VMD 90 μm) and Low Drift (VMD 276 μm) nozzles, compared to background levels from either nozzle alone. The Conventional nozzles produced 55% more larger fine droplets, while the Low Drift nozzles generated a higher quantity of fine particles (<1.0 μm) at distances of 25 and 30 m from the sprayer (Grella et al., 2023). Recent advancements in drone and helicopter technology have made these aerial applications increasingly common, offering enhanced precision in targeting and the ability to cover larger areas efficiently (Hafeez et al., 2023). However, prioritizing ground application over aerial application whenever possible could help minimize drift incidence. In a comparative study, aerial application resulted in 5–8 times greater herbicide drift compared to ground application under identical weather conditions (Butts et al., 2022).
- **Buffer zones**: Establishing buffer strips of vegetation between farmland and surrounding areas can significantly reduce drift. Studies show that wider buffer zones, particularly those of 8 m or more offer great protection against drift (Lorenz et al., 2022). These designated areas between farmland and sensitive environments (or non-target areas) act as physical barriers, intercepting and reducing the movement of airborne pesticides. The benefits of these strips, however, must be adopted in combination with other protective measures.
- **Hedgerows**: hedgerows serve as natural barriers against pesticide drift. When used and maintained properly, hedgerows act by filtering pesticide spray, reducing herbicide drift up to 80% (Boutin et al., 2014; Lazzaro et al., 2008). Their effectiveness is influenced by several factors, including canopy density, height, structure and wind speed during pesticides applications (Connor et al., 2014). However, their performance can vary seasonally, particularly for deciduous hedgerows, which could be less effective during leaf-off periods. For instance, Linhart et al. reported peak seasonal drift rates in spring and summer in South Tyrol, Italy, with 76% of detected residues being endocrine active. The highest concentrations were for chlorpyrifos-methyl (0.71 mg/kg), oxadiazon (0.64 mg/kg), captan (0.46 mg/kg), and fluazinam (0.23 mg/kg). Residue quantity, concentration, and contamination rates differed by season ($p < 0.001$): in spring, 25 residues appeared at 83% of sites (median 0.240 mg/kg); in summer, nine residues at 79% (0.092 mg/kg); in autumn, three at 50% (0.076 mg/kg); and in winter, four at 17% (0.155 mg/kg) (Linhart et al., 2021). Therefore, dense, multi-layered hedgerows with a mix of deciduous and evergreen species tend to be most effective at intercepting drift. However, this comes at a potential cost. Trapped pesticides within the hedgerow foliage can negatively impact beneficial arthropods like pollinators and predators residing there, and can act as barrier effects on pollination services (Klaus et al., 2015; Otto et al., 2009). This highlights the complex role of hedgerows in orchard ecosystems.
- **Standardized monitoring**: Developing flexible protocols that enable local adaptability while ensuring robustness is essential for effective biodiversity monitoring. This can be achieved by utilizing Essential Biodiversity Variables (EBVs) as a foundational framework that guides both data collection and analysis (Hoban et al., 2022; Silva del Pozo et al., 2023). The EBV framework facilitates the identification of key indicators, assessment of site-specific characteristics, and allocation of resources needed to support and promote the development of comprehensive biodiversity monitoring programs. Subsequently, a parallel data workflow must be developed to produce and make both EBV and raw data freely accessible, which can help in integrating results across various biodiversity monitoring communities (Schäfer et al., 2019; Sylvester et al., 2023). Furthermore, European Food Safety Authority (EFSA) advocated for harmonizing OECD Test Guidelines and Guidance Documents in the EU pesticide regulatory framework and highlighting uncertainties.
- **Improved ERA**: The need for refined ERA models to improve ecological models used for impact prediction, and to develop frameworks that are capable of assessing the combined effect of multiple stressors on biodiversity (Dalton et al., 2023; Schmeller et al., 2017). An important issue is that current testing methods primarily rely on crop plants instead of the wild plants actually at risk, and focus on short-term effects in artificial settings. This approach casts doubt on whether current test choose the right effect indicators (Boutin et al., 2014; Damgaard et al., 2014). Thus, metrics such as monitoring flowering rates in wild plants may be a more realistic indicator of ecosystem health than simply counting plant species (Strandberg et al., 2019).
- **Potential use of drift retardants**: Drift retardants can reduce the risk of drift by increasing droplet size, thereby reducing the potential for spray drift. A study conducted by Hoffmann et al. found that some polymer-based drift retardants were able to reduce the risk of drift by up to 70% compared to water alone (Hoffmann et al., 2008). Another study by Nuyttens et al. found that the volume of driftable fine droplets was reduced by 75–90% when drift retardants were used

(Foqué and Nuyttens, 2011). However, if too high a concentration of drift retardants is applied, the potential for drift increases rather than decreases, as very large droplets are formed that shatter on impact (Hewitt, 2000). In addition, there has always been concern that the interaction between these chemicals and pesticides would reduce the effectiveness and reach of the pesticides (Song et al., 2022; Zhao et al., 2022). Therefore, the use of drift retardants must be carefully calculated and in accordance with the instructions of the manufacturers or authorized agencies. Fig. 5 summarizes currently available strategies to mitigate pesticide drift.

Finally, a Europe-wide study suggests that banning harmful pesticides alone is not enough, and that a paradigm shift towards biodiversity-based solutions like biological pest control is crucial to truly mitigate pesticide damage (Geiger et al., 2010).

6. Conclusions and recommendations for future research

Pesticide drift represents a multifaceted challenge with profound consequences for biodiversity and ecosystem health. The evidence presented in this review suggests that the harmful effects of airborne pesticides extend well beyond agricultural fields, disrupting ecological balances and endangering species critical to ecosystem services. This situation is, however, exacerbated by significant knowledge gaps resulting from a lack of published research, uncoordinated data collection across regions, and a limited understanding of how factors such as land use, weather, and pesticide properties influence airborne pesticide levels and their effects on non-target organisms.

The negative environmental impacts of pesticide drift are evident at every level of ecosystems and biology. Impacts include reduced reproductive rates, changes in growth, development, and/or behavior, modification of diversity or community organization, disruption of food webs, and declines of important species. Pesticides disrupt the delicate balance between species that define a functioning ecosystem. Impacts can be local, transnational, or even continental.

Obviously, it is difficult to know for sure what the consequences of pesticide drift might be. Given this, the logical and safest course of action is to adopt a precautionary strategy by phasing out the most harmful pesticides, reducing our reliance on toxic pesticides, and advocating environmentally sustainable pest management. One successful idea in this regard is integrated pest management. This approach involves the minimal use of the least harmful pesticides, combined with biological and cultural strategies to reduce pest losses. It involves using pesticides only when pest attacks exceed threshold levels.

In addition, since organic agriculture completely prohibits the use of pesticides, it can be considered one solution to reducing the problems of pesticide drift. Encouraging the consumption of organic foods would motivate farmers to choose organic farming. Market dynamics act as a strong incentive for farmers to adopt organic practices. Pesticide drift is a global issue that calls for collective action to regulate pesticide use to minimize negative environmental consequences.

The documented risks of pesticide drift underscore the urgent need for increased funding of this area of research to improve our understanding of pesticide drift mechanisms and develop safer, more sustainable alternatives for pest control. The impact of airborne pesticide contamination is often not acute, which means that its observation is limited to long-term, in-depth studies. Thus, intensified research studies are needed to gain insight into how to conduct tests and collect relevant information on the environmental issues caused by pesticides. One challenge to such studies is that we lack a control group exposed to pesticides as a reference. For this reason, we may not be able to measure the exact risks of these exposures. Increasing financial support from independent sources is critical to fill existing knowledge gaps, encourage long-term studies, and support innovative practices that prioritize environmental health over commercial interests. Moreover, long-term studies are essential to understand recovery dynamics and ecosystem services after exposure, but when such studies are not possible, extrapolation methods must be used to gain insights. Another challenge is the lack of standardized monitoring methods and metrics for pesticide deposition and volatilization complicates accurate risk assessments, making the development of consistent methods essential. To effectively address these challenges, scientific oversight and stakeholder involvement are essential.

Fig. 5. Possible strategies and methods to reduce pesticides drift.

Improved risk assessment methods must consider the cumulative impacts of pesticide drift and require comprehensive studies that examine both direct and indirect impacts on biodiversity.

To move forward, policymakers, researchers, and agriculture stakeholders must work together to develop innovative solutions that consider the wide-ranging impacts of pesticide drift. Efforts to mitigate these impacts should be complemented by a commitment to regulatory frameworks that protect biodiversity and ecosystem integrity. Bridging the gap between toxicology and sustainable agricultural practices is critical to developing effective, environmentally sound pest management strategies.

Finaly, a concerted effort to encourage collaboration will empower research that provides a clearer picture of the impacts of aerial pesticides on biodiversity. Adopting comprehensive strategies to address pesticide drift can preserve biodiversity for future generations and ensure a resilient and thriving environment.

CRediT authorship contribution statement

Saeed S. Albaseer: Writing – review & editing, Writing – original draft, Visualization, Conceptualization. **Veerle L.B. Jaspers:** Writing – review & editing, Conceptualization. **Luisa Orsini:** Writing – review & editing, Conceptualization. **Penny Vlahos:** Writing – review & editing. **Hussein E. Al-Hazmi:** Writing – review & editing. **Henner Hollert:** Writing – review & editing, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Saeed Albaseer reports financial support was provided by Alexander von Humboldt Foundation. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

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