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Passerine birds as bio-indicators of pesticide levels in farmlands and gardens: mitigation by organic farming and distance from treated fields

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ABSTRACT

Both farmworkers and residents living near agricultural areas face adverse health outcomes prevailingly attributed to pesticides. Proximity to intensive agricultural areas may increase exposure, although this association remains debated, partly because collecting human biological samples is difficult. Therefore, identifying relevant bioindicators of environmental pesticide exposure could provide an alternative approach, and passerine birds are promising candidates given their close coexistence with humans. Here, we compared pesticide levels and profiles in passerines from farmland hedgerows and residents' gardens within the same rural area, and explored the effects of two mitigating components, assuming lower pesticide pressure with a higher percentage of organic farming (%OF) around hedgerows and with greater distance from the nearest treated crop for gardens. Blood samples were collected from 217 individuals across 11 species, and 111 pesticide residues were analysed. Contamination levels and profiles did not differ between farmlands and gardens, suggesting pervasive pesticide occurrence driven by environmental persistence and broad-scale dispersal. In farmland, model-estimated pesticide concentrations declined by 61.3 % (95 % CI: 32.5–77.8 %) as %OF increased from 0 to 100 %. In gardens, the model-estimated number of compounds and pesticide concentrations decreased with distance to the nearest conventional plot by 30.1 % (95 % CI: 6.8–47.6) and 70.1 % (95 % CI: 49.2–82.4), respectively. These effects were species-dependent, suggesting that certain species, such as the European robin, are more relevant indicators of local pesticide levels, likely due to their smaller home ranges. By showing that bird contamination reflects local agricultural practices and proximity, this study supports using passerines as effective sentinels of local environmental pesticide pressure, offering a valuable proxy for assessing ecological exposure in environments shared with human populations.

1. Introduction

Pesticides are among the most widespread anthropogenic contaminants in the environment. They are detected across virtually all land-use types and compartments, from intensively farmed landscapes to urban environments (Md Meftaul et al., 2020) and remote ecosystems far removed from any application source (Brühl et al., 2024; Guida et al., 2018; Mayer et al., 2024; Wang et al., 2022). This widespread environmental dispersal of pesticides from treated crops to surrounding environments is largely driven by rainwater run-off (Bianco et al., 2025; Chen et al., 2019) and spray drift (Felsot et al., 2010; Lazzaro et al., 2008). Such pervasive contamination affects a wide range of organisms, including wildlife and humans, beyond the pests, weeds and fungi originally locally targeted within crops. Agricultural intensification, and pesticide use in particular, is now recognised as one of the main drivers of biodiversity decline across taxa, and especially for birds inhabiting farmland environments (Chamberlain et al., 2000; Geiger et al., 2010; Maxwell et al., 2016; Rigal et al., 2023; Stanton et al., 2018). These impacts operate both indirectly, through habitat loss and fragmentation (Butet et al., 2010; Fischer et al., 2011; Martin et al., 2020) and reduced

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abundance and diversity of food resources (Blackburn and Wallace, 2001; Boatman et al., 2004; Girard et al., 2014; Sánchez-Bayo, 2021), and directly, *via* acute exposure causing lethal effects (Addy-Orduna et al., 2019; Millot et al., 2015; Poliserpi et al., 2021) or chronic low-dose exposure causing sublethal effects (Gibbons et al., 2015; Moreau et al., 2022a; Wan et al., 2025).

Nonetheless, agricultural pesticide pressure is not uniform: it varies with farming practices. Organic farming has emerged as an alternative agricultural model that addresses these environmental concerns by working within natural ecological balances and prohibiting synthetic inputs in favour of organic fertilisers (Lockeretz, 2007; Van Mansvelt et al., 1998). This results in higher abundance, richness and diversity of multiple taxa in organically managed farmland, including plants (Hyvönen et al., 2003; Rundlöf et al., 2010), insects (Gabriel and Tscharntke, 2007; Grabovska et al., 2020) and birds (Belfrage et al., 2005; Chamberlain et al., 2010; Katayama, 2016; McKenzie and Whittingham, 2009; Smith et al., 2010). Organic farming therefore appears a promising strategy for mitigating biodiversity loss in temperate agricultural landscapes (Stein-Bachinger et al., 2021). The prohibition of synthetic inputs, the only mandatory regulatory requirement, directly reduces environmental contamination: soils under organic management in Europe, for instance, show markedly lower pesticide residue concentrations than conventionally farmed soils (Geissen et al., 2021). This reduced pressure also extends to wildlife, with lower pesticide contamination levels correspondingly reported in avian species under organic management (e.g., Bailly et al., 2025; Fuentes et al., 2024a). Conversely, proximity to intensively cultivated fields has been associated with increased pesticide exposure, both through direct quantification in environmental matrices (e.g., household dust and indoor and outdoor air, Dereumeaux et al., 2020; Deziel et al., 2017; Teysseire et al., 2021; Wofford et al., 2014) and biological samples (e.g., urine, blood, and hair, Chevrier et al., 2014; Mercadante et al., 2013; Van Wendel De Joode et al., 2012; Wu et al., 2013). For instance, air samples collected from proximal households (< 250 m from crops) showed significantly higher concentrations of chlorpyrifos, chlorpyrifos-oxon, and azinphos-methyl compared to those from more distant households (Gibbs et al., 2017). Yet, while this proximity effect is well documented for pesticides in human exposure studies, comparable evidence for pesticides in wildlife remains scarce. Exposure studies in wildlife have mostly focused on aquatic or agricultural habitats, rarely testing this proximity gradient for pesticides in non-agricultural, and in particular urban or peri-urban, environments. Moreover, most studies remain limited to a small number of compounds or specific chemical families and therefore fail to capture the complexity of real-world pesticide mixtures (Dereumeaux et al., 2020; Moreau et al., 2022a). Assessment of pesticide contamination to a larger number of compounds and/or chemical families simultaneously among both agricultural and non-agricultural areas is therefore essential to better understand the links between pesticide contamination and both agricultural practices and the proximity to treated crops, and to improve the accuracy and reliability of studies on pesticide effects on non-targeted organisms.

Bioindicator species offer a complementary approach to characterise pesticide exposure: by integrating contamination across multiple environmental compartments, they reflect the bioavailable fraction of contaminants that organisms actually encounter in their habitat (Hamza-Chaffai, 2014; Siddig et al., 2016). Birds are particularly recognised as effective bioindicator species for monitoring environmental contaminants and their biological effects, owing to their broad geographic distribution and ecological diversity (Becker, 2003; Mo et al., 2018). For example, biomonitoring of several Arctic bird species highlighted increased mercury contamination in the Arctic (Berglund, 2018). High levels of pesticide contamination have also been reported in several bird species (Yohannes et al., 2017), including farmland species such as partridges (Bariod et al., 2025; Millot et al., 2015), raptors (Byholm et al., 2018; Fuentes et al., 2023, 2025) and passerines (Angelier et al., 2023; Bailly et al., 2025; Humann-Guilleminot et al., 2019; Tank et al., 2026). Passerines are distributed worldwide and rank among the most abundant birds in almost all terrestrial environments. This order comprises the greatest species diversity among birds, with an estimated 6,500 species worldwide (Schmitt and Edwards, 2022) and ca. 245 species in France, either resident or migratory (breeding or passage). This taxonomic and ecological diversity, encompassing variation in biology, foraging ecology, diet and habitat preference, allows passerines to monitor a broad range of environments and biodiversity components. This interspecific variability may, in turn, shape pesticide exposure (e.g., Bailly et al., 2025), as bioaccumulation and biomagnification are jointly influenced by a wide range of factors, including species-specific physiology and biology (Kuo et al., 2022), encompassing differences in detoxification capacity, foraging strategy and mobility that can strongly affect pesticide uptake and metabolism (Cooper et al., 2017; Lucia et al., 2012; Rainio et al., 2012). Resident passerines, in particular, are especially suitable for assessing local contamination due to their small home ranges and foraging areas during the breeding season (Yu et al., 2014), while migratory species may reflect broader-scale or cumulative exposure across their range. In this sense, passerines can serve as indicators of the local environmental exposome, enabling comparisons of pesticide exposure levels across environments that differ in land use and pesticide pressure.

The present study aimed to assess pesticide contamination in wild passerine birds and compare contamination levels and profiles between two contrasted habitats: agricultural areas, represented by farmland hedgerows, and non-agricultural areas represented by residents' gardens in villages. We quantified in the blood of eleven passerine species 111 pesticide compounds, including fungicides, herbicides, insecticides and additives. A key challenge in assessing pesticide exposure in wildlife is that truly unexposed "control" areas no longer exist. Exposure can therefore only be meaningfully assessed by contrasting environments along gradients of pesticide use, comparing areas subject to different levels of pesticide pressure. We further explored the effect of pesticide pressure on bird pesticide contamination, by testing the effects of two potential mitigating components on pesticide levels: (1) the increased percentage of crops in organic farming (%OF) in farmland birds; and (2) the increased distance from treated crops in garden birds. We therefore selected hedgerows along a gradient of percentage of area in organic farming (%OF), and we expected that birds caught in hedgerows with higher %OF would show lower pesticide levels than birds in hedgerows with a higher percentage of area in conventional farming. In residents' gardens, we tested the hypothesis that birds caught in gardens bordering conventional fields would have higher contamination levels than birds caught in gardens far from conventional fields by selecting gardens along a gradient of distance to the nearest conventional field. Overall, we expected birds caught in farmland hedgerows to have higher contamination levels than those in gardens because agricultural fields have higher local application rates than residential areas, where pesticide use is strictly limited or prohibited.

2. Materials and methods

This study complies with French legal regulations and ethical standards. Capture permits were provided by the Centre for Research on the Biology of Bird Populations (CRBPO, National Museum of Natural History, Paris) and all procedures were approved by the Committee of Animal Experimentations of the Deux-Sèvres French District (APAFIS#32899-2022022512188344).

2.1. Study area and site selection

This study was conducted within the Long-Term Socio-Ecological Research site "Zone Atelier Plaine & Val de Sèvre" (ZAPVS), located in the south-west of France (46°11′ N, 0°28′ W). The ZAPVS covers 435 km^2 and includes 24 municipalities that are entirely within the ZAPVS and eight others partially within. It is an intensive farming area with diverse

farming systems and landscapes (Bretagnolle et al., 2018). Winter cereals represent the main crops (41.5% on average from 2009 to 2016), followed by wheat (33.8%), sunflower (10.4%), maize (9.6%), oilseed rape (8.3%), and peas (2%). Meadows (including both permanent grasslands and temporary hay) account for about 13.5% of the land area, whereas urban areas cover 9.8% of the surface area of the ZAPVS (Bretagnolle et al., 2018). The land cover, including the location of organic farmed fields, have been monitored annually since 1994, allowing mapping each field in organic farming, which covered ~18% of the agricultural area and ~80 farms in 2022, while ~350 farms are in conventional farming.

Within the ZAPVS, 31 sampling sites were selected in 2023 as follows. Firstly, 12 farmland hedgerows at the edge of crops were selected according to a gradient of the percentage of fields in organic farming within a 250-meter radius buffer (Fig. 1). The home range sizes during the breeding season for passerine species range from 0.2 ha to 5 ha, which correspond to circular territories within a radius comprised between 20 and 150m approximately (Assandri et al., 2017; Collar and Christie, 2020; Ferry et al., 1981; Kumstátová et al., 2004; Morganti et al., 2017; Tuomenpuro, 1989; Vangestel et al., 2010). We therefore chose a radius of 250m to have an additional margin and assume that birds cannot exceed the buffer. The selected hedgerows were graded between 0% and 100% organic farming within the buffer (Table 1). The structural characteristics of the hedgerows, such as length, height, density, and thickness, did not differ between the farming systems, according to the methodology used in a previous study (Moreau et al., 2022b). Secondly, 19 domestic gardens were selected in two municipalities (Fig. 1): BSN and FO (names of municipalities are confidential), which have a similar percentage of organic farming within a 1000m radius buffer around the centroid of the village (~22% organic farming around BSN and ~18% organic farming around FO) though it differs more at 2000m radius buffer (28% and 12% respectively). In each municipality, we surveyed the gardens of local partners who volunteered to participate in the study (see Gaba et al., 2026 for details on residents' recruitment). From these, we selected 10 gardens in BSN and 9 gardens in FO (whose GPS coordinates are confidential) based on predefined criteria. Eligible gardens had to be sufficiently large and designed to be "bird-friendly", including mandatory requirements (e.g., presence of hedges and lawns) as well as optional features (e.g., vegetable gardens, water sources). The most important selection criterion for gardens was their distance to the nearest treated field to ensure a gradient of distance to the nearest conventional field (Table 1).

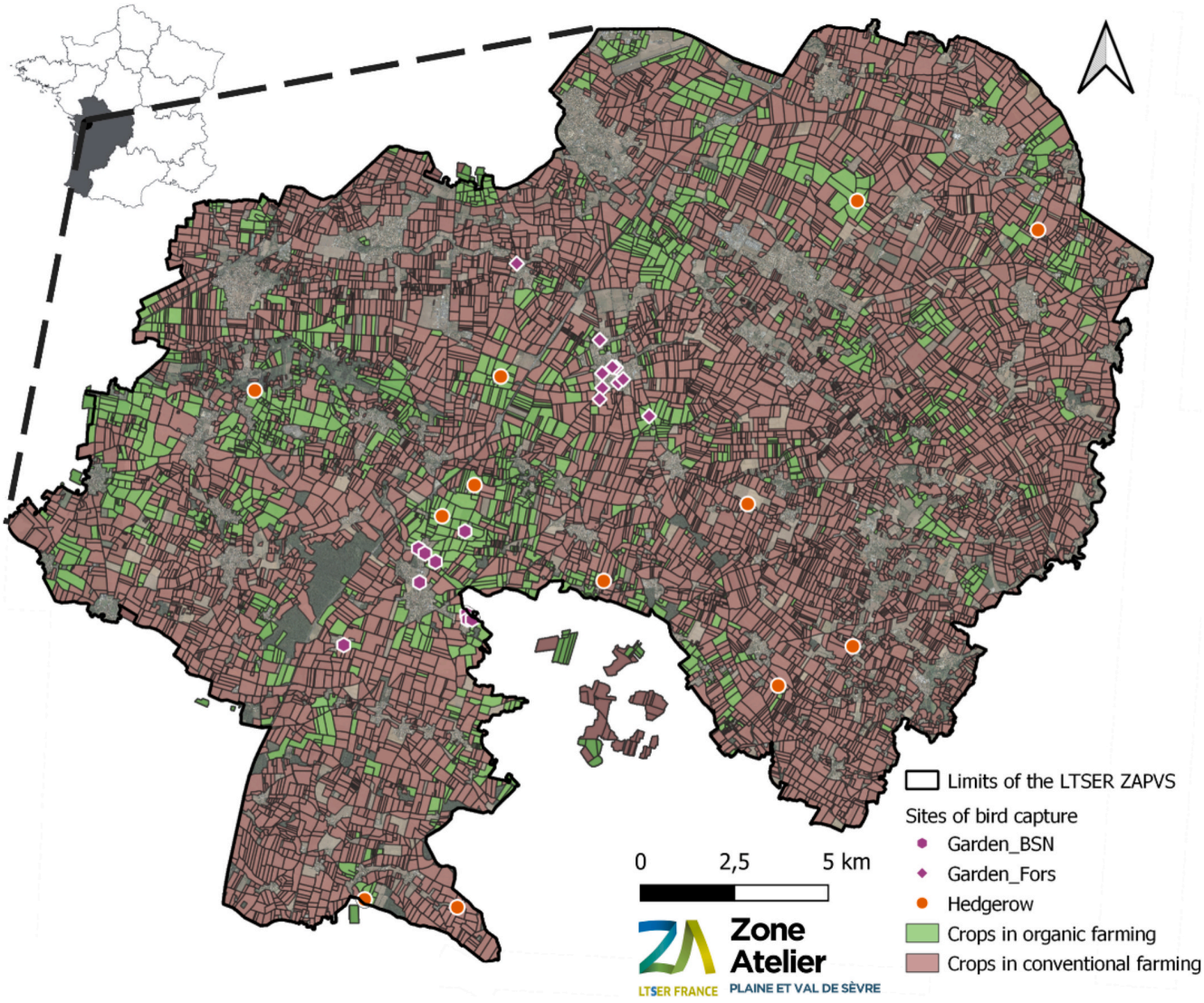


Fig. 1. Location of the 31 sampling sites in the Zone Atelier Plaine & Val de Sèvre (ZAPVS). Crops in organic farming appear in green, and crops in conventional farming appear in brown. Orange circles represent farmland hedgerows, and hexagons and squares in purple show gardens selected at BSN and FO, respectively.

Table 1
Characteristics of the 31 sampling sites, with the percentage of organic farming in a 250m radius buffer for farmland hedgerows, and the distance to the nearest conventional crop (Dist. to nearest conv. crop) for sampling gardens in both municipalities (i.e., BSN and FO). The types of crops cultivated in the year of the study (2023) are given for farmland hedgerows and gardens.

Type of habitat	ID Site	% OF	Dist. to nearest conv. crop (m)	Type of crop
Farmland hedgerows				
	H 1	0.00	-	Sunflower / Wheat
	H 6	0.00	-	Wheat
	H 7	0.00	-	Maize
	H 3	5.69	-	Sunflower / Wheat
	H 4	10.06	-	Wheat
	H 5	17.23	-	Maize / Sunflower
	H 11	25.82	-	Sunflower
	H 9	63.51	-	Wheat
	H 12	77.88	-	Sunflower
	H 10	88.38	-	Wheat
	H 2	89.50	-	Sunflower / Peas
	H 8	100.00	-	Sunflower / Wheat
Domestic gardens				
BSN	BSN 13	-	0	Barley
	BSN 32/33	-	107	Barley
	BSN 20	-	135	Wheat
	BSN 16	-	139	Peas
	BSN 9	-	165	Oilseed rape
	BSN 3	-	167	Sunflower
	BSN 23	-	178	Wheat
	BSN 35	-	288	Oilseed rape
	BSN 24	-	298	Wheat
FO	FO 3	-	0	Flax
	FO 14	-	0	Wheat
	FO 4	-	10	Wheat
	FO 15	-	103	Ray grass
	FO 12	-	162	Barley
	FO 11	-	191	Wheat
	FO 10	-	242	Wheat
	FO 7	-	248	Sunflower
	FO 13	-	344	Sunflower
	FO 6	-	378	Sunflower

Proximity to the conventional field was measured in QGIS 3.22.16 from the centroid of the garden to the edge of the nearest conventional crop, based on shapefiles of the land cover from 2023 and shapefiles of organic farming inventoried in 2022 in the ZAPVS. These nearest conventional crops mostly consisted of large agricultural patches exceeding six hectares, thus representing significant potential sources of pesticide pressure. Only one garden (BSN 16) was located near a smaller, 2.5-hectare isolated patch. The proximity to the nearest conventional crop ranged from 0m to 298m in BSN and 0m to 378m in FO.

2.2. Bird captures and blood collection

Bird captures were conducted in each habitat (hedgerows and gardens) using the same method (net trapping) and conditions (start and end times of capture, weather conditions) as described in Moreau et al. (2022b). As there were not enough juveniles caught, this study focuses solely on adults. Males and females were captured and identified by their plumage and reproductive characteristics (i.e., cloacal protuberance or brood patch). When these elements were not visible, the individuals were sexed molecularly by polymerase chain reaction (PCR) on whole blood samples at the Centre d'Etudes Biologiques de Chizé. Bird captures were made during two sessions at each site during the 2023 breeding season, from April 5th to June 3rd for the first session, and from June 5th to July 20th for the second one. Overall, 217 adult birds from 11 species were captured and identified with a numbered metal ring from the French National Museum of Natural History, Centre de Recherches sur la Biologie des Populations d'Oiseaux (MNHN Paris, CRBPO). A few individuals were recaptured across sessions, but only data from the first capture of each individual were retained for analysis, and any subsequent recapture was excluded, so that each bird contributed a single blood sample to the dataset. Only species with enough individuals (>5) caught in hedgerows and in gardens were included in analyses (Table 2): seven species were retained for statistical analyses in hedgerows, nine species were kept for statistical analyses in gardens, and five species commonly caught in both habitats were included in the comparison between them. Blood samples were collected from each bird by puncturing the brachial vein with a sterile needle and heparinised capillaries. As blood reflects recent pollutant contamination (Espín et al., 2016) and passerine birds have small territories during the breeding season, this matrix can be used to assess local exposures. Blood samples of 50 μL were placed in Eppendorf tubes at 0–5 °C and then stored at the laboratory at −20 °C for subsequent analyses.

2.3. Multi-residue analyses of pesticides

The blood samples were analysed by the Institut de Chimie des Procédés pour l'Energie, l'Environnement et la Santé (UMR 7515, ICPEES) at the University of Strasbourg (France). Multi-residue analyses allowed detection and quantification in whole blood samples of a list of 111 pesticide compounds among the most used in France. The method was developed and detailed in Rodrigues et al. (2023) and used in several previous studies (Bailly et al., 2025; Bariod et al., 2025; Fuentes et al., 2023, 2024a, 2024b, 2025). Two analytical methods were implemented: liquid chromatography coupled with tandem mass spectrometry (LC-MS/MS) for the least volatile compounds and automated thermal desorption gas chromatography coupled to tandem mass spectrometry (ATD-GC-MS/MS) for the most volatile compounds (Rodrigues et al., 2023).

The list of 111 compounds (Table S1 in Supplementary Information) included 39 fungicides, 40 herbicides, 29 insecticides, 2 safeners (improving the herbicide selectivity towards weeds) and one synergist (enhancing pesticide efficacy). The use of these compounds varies, with

Table 2
Summary of descriptive parameters by bird species and type of habitat (hedgerows, gardens in BSN and in FO) in which birds were caught. The number of birds caught ("Nb.") in each type of habitat is given for the species retained for analyses in the hedgerows in orange and for analyses in the gardens in purple. Species in bold are those retained in the comparison between both environments. The mean N_C and C_T correspond to the mean number of pesticides detected and the mean total concentration (in pg.mg^{-1}) per individual, with their standard deviation (sd). The range (minimum – maximum) corresponds to the lower and upper values found for an individual.

Bird species	Type of habitat	Nb.	Mean N_C ± sd	Range	Mean C_T ± sd	Range
Black redstart	Hedgerows	0	-	-	-	-
	Gardens in BSN	9	5.00 ± 3.28	2-12	475.63 ± 356.78	67.44-1090.72
	Gardens in FO	8	4.50 ± 1.41	2-7	594.89 ± 478.93	63.37-1516.99
Common blackbird	Hedgerows	17	4.47 ± 1.66	2-7	312.27 ± 292.99	11.43-843.09
	Gardens in BSN	27	4.37 ± 2.32	1-10	621.71 ± 712.15	8.57-3225.77
	Gardens in FO	27	4.56 ± 2.26	1-10	324.28 ± 315.67	0.37-1005.94
Common linnet	Hedgerows	2	4.50 ± 0.71	4-5	309.95 ± 63.32	264.97-354.94
	Gardens in BSN	2	5.50 ± 0.71	5-6	130.55 ± 41.60	101.14-159.97
	Gardens in FO	4	6.25 ± 1.71	4-8	508.92 ± 279.43	347.43-925.33
Common nightingale	Hedgerows	8	3.88 ± 1.55	2-6	767.38 ± 689.76	11.90-2084.56
	Gardens in BSN	1	3.00	-	714.14	-
	Gardens in FO	1	4.00	-	227.00	-
Dunnock	Hedgerows	6	5.83 ± 5.64	1-17	395.83 ± 234.13	68.16-751.97
	Gardens in BSN	3	5.00 ± 2.65	2-7	607.69 ± 686.06	36.79-1368.79
	Gardens in FO	4	4.25 ± 1.26	3-6	462.04 ± 623.12	49.03-1385.79
Eurasian blackcap	Hedgerows	16	5.50 ± 3.18	1-14	583.30 ± 618.30	7.09-2038.82
	Gardens in BSN	9	5.78 ± 3.56	2-13	365.94 ± 287.92	35.44-876.68
	Gardens in FO	12	5.58 ± 3.82	1-12	452.02 ± 518.81	24.53-1583.63
Eurasian chaffinch	Hedgerows	3	3.33 ± 2.31	2-6	285.83 ± 427.42	20.89-778.91
	Gardens in BSN	1	5.00	-	199.10	-
	Gardens in FO	8	5.75 ± 2.19	2-9	556.88 ± 263.56	69.99-856.73
European goldfinch	Hedgerows	0	-	-	-	-
	Gardens in BSN	3	5.00 ± 1.73	4-7	165.48 ± 102.06	49.71-242.47
	Gardens in FO	4	4.75 ± 2.22	3-8	427.67 ± 330.08	64.43-820.47
European robin	Hedgerows	5	3.80 ± 1.64	2-6	242.98 ± 258.46	37.04-677.46
	Gardens in BSN	2	3.50 ± 3.54	1-6	163.35 ± 8.06	157.65-169.06
	Gardens in FO	6	7.17 ± 2.86	2-10	693.79 ± 449.18	26.39-1191.53
Great tit	Hedgerows	6	4.67 ± 1.51	3-7	541.24 ± 546.59	63.99-1482.21
	Gardens in BSN	4	6.75 ± 0.50	6-7	959.23 ± 1136.56	181.09-2605.19
	Gardens in FO	8	7.25 ± 3.45	2-13	511.37 ± 354.80	137.03-1037.36
Melodious warbler	Hedgerows	9	5.33 ± 2.50	1-10	490.14 ± 629.32	37.78-1845.45
	Gardens in BSN	1	7.00	-	113.28	-
	Gardens in FO	0	-	-	-	-

55 compounds for farming use only, 54 for farming and domestic use (compounds which can be found in urban environments, used on non-cropped and utility sites, including buildings, cemeteries, stored products, sport fields, ornamental plants, lawns, turf for instance) and two for farming, domestic and veterinary use (for domestic pets, nitenpyram and imidacloprid). Some chemical properties of each compound are summarised in Table S1, based on data retrieved from the Pesticide Properties DataBase (PPDB, at https://sitem.herts.ac.uk/aeru/ppdb/en/atoz.htm, Lewis et al., 2016). This database compiles ecotoxicological information for all compounds included in this study and has been widely used in previous research to obtain parameters such as DT_{50} and LD_{50} values (e.g., Froger et al., 2023; Pelosi et al., 2021). Limits of detection (LODs) and quantification (LOQs) for each compound are provided in Table S1 and were validated using spiked blank samples. Analytical blanks (to guarantee data quality) were processed in parallel with the samples under identical conditions, using the same analytical methods, and no contamination was detected.

We assessed pesticide contamination levels through three parameters commonly used in ecotoxicological studies: the number of compounds detected ("N_C" hereafter), the total concentration ("C_T" hereafter), and the total toxicity index ("TI" hereafter). The number of compounds corresponds to the total number of pesticide compounds detected (out of the 111 sampled) in an individual and reflects the chemical diversity of contamination. The total concentration was calculated as the sum of the concentrations of each detected compound. Each blood sample was dried and weighed before extraction to normalize the results to the exact mass of each sample, given the difficulty of determining an exact volume in whole blood samples, but therefore not accounting for the heterogeneity of blood volume among species and individuals. Concentrations were expressed as pg.mg^{-1} (picograms of analyte per milligram of dried blood). Finally, the total toxicity index was computed as the sum of each division of the concentration of each compound by its respective LD_{50} value (obtained from the Pesticide Properties Database PPDB at https://sitem.herts.ac.uk/ae

ru/ppdb/en/atoz.htm; Lewis et al., 2016). This toxicity index approach carries several limitations, as it assumes additive toxicity despite differing modes of action of compounds. Moreover, LD_{50} values, although the only toxicity metric available for all detected compounds, reflect acute rather than chronic exposure. Blood concentrations reflect internal exposure and are not directly equivalent to the orally administered doses used to derive LD_{50} values. Moreover, the available LD_{50} values were primarily derived from Japanese quail and mallard and may not reflect the sensitivity of the passerine species sampled here. Because species-specific toxicity data were unavailable, the index could not account for interspecific differences in sensitivity. The TI should therefore not be interpreted as a direct quantitative estimate of toxicological risk; it was used only as a relative, toxicity-weighted metric complementary to the number of compounds and total pesticide concentration.

2.4. Statistical analysis

In the present study, we explored pesticide contamination in passerine birds in three steps: (1) the comparison between birds caught in farmland hedgerows and those caught in gardens; (2) the effect of organic farming on contamination levels for birds caught in hedgerows; and (3) the effect of the proximity to conventional crops on contamination levels for birds caught in gardens. All statistical analyses were performed in R v.4.2.2 software (R Core Team, 2022).

2.4.1. Comparison of pesticide contamination between habitats

To compare pesticide contamination levels between habitats, five species commonly caught in both farmland hedgerows and residents' gardens were included: the Common Blackbird (*Turdus merula*), Dunnock (*Prunella modularis*), Eurasian Blackcap (*Sylvia atricapilla*), European Robin (*Erithacus rubecula*), and Great Tit (*Parus major*). Differences were investigated using a Mixed-Generalised Linear Model fitted with a Negative Binomial distribution (NB-GLMM) for N_C, a Gamma distribution (G-GLMM) for C_T and a linear model (LMM) for $\log_{10}$-transformed TI. Explanatory variables included habitat type (hedgerows, gardens in BSN, gardens in FO), species, their interaction, sex, and capture date, and the site of capture was added as a random variable. Capture date was included as a quadratic term in the total pesticide concentration (C_T) model. Pairwise contrasts among habitat types were calculated from estimated marginal means using the "emmeans" package (Lenth, 2017), with 95% confidence intervals and Tukey adjustment for multiple comparisons. To account for potential spatial structure in the data, we georeferenced each site using its coordinates (Lambert 93, EPSG:2154). Spatial autocorrelation was assessed on model residuals using the "DHARMa" package (Hartig et al., 2024), applying the *recalculateResiduals* and *testSpatialAutocorrelation* functions, which implement a distance-based Moran's I test. Across all models, observed Moran's I values were close to zero and non-significant ($p > 0.05$), indicating no detectable residual spatial structure among sites beyond what was already accounted for by the fixed and random effects included in each model (i.e., habitat type and site of capture). This suggests that residual spatial dependence among hedgerows and gardens was not a concern for the analyses presented here.

We also analysed pesticide contamination profiles (compositional diversity) across habitats and species by assessing beta diversity using the Jaccard index, computed on the presence/absence of detected pesticide compounds. Then, we performed a permutational multivariate analysis of variance (PERMANOVA) based on Jaccard dissimilarities, with 1000 permutations, using the *adonis2* function from the "vegan" package (Oksanen et al., 2025). This analysis showed how habitat type (i.e., hedgerows, gardens in BSN, and gardens in FO) and species (five species commonly caught in both environments; Table 2) influence the presence of pesticide compounds. To visualise this analysis, we also represented pesticide contamination profiles using non-metric multidimensional scaling (NMDS) based on the same Jaccard dissimilarity matrix, with ellipses showing sample dispersion within each habitat type.

2.4.2. Evaluation of two potential mitigating components: organic farming and distance from the nearest conventional field

We evaluated two potential mitigating components through separate analyses. To explore the effects of the percentage of organic farming in a 250m radius buffer, we focused on birds caught in farmland hedgerows, including seven species: the Common Blackbird, Common Nightingale (*Luscinia megarhynchos*), Dunnock, Eurasian Blackcap, European Robin, Great Tit and Melodious Warbler (*Hippolais polyglotta*) (Table 2). To explore the effects of the distance from the nearest conventional field, we included nine species caught in residents' gardens: the Black Redstart (*Phoenicurus ochruros*), Common Blackbird, Common Linnet (*Linaria cannabina*), Dunnock, Eurasian Blackcap, Eurasian Chaffinch (*Fringilla coelebs*), European Goldfinch (*Carduelis carduelis*), European Robin and Great Tit (Table 2). To assess how species composition was distributed across both gradients, we ran PERMANOVA using Jaccard dissimilarities (presence/absence data) and Bray-Curtis dissimilarities (abundance data), testing each gradient separately: the percentage of organic farming in the surrounding landscape and distance from the nearest conventional field. Neither species composition (Jaccard - %OF: $P = 0.61$; distance: $P = 0.46$) nor abundance profiles (Bray-Curtis - %OF: $P = 0.52$; distance: $P = 0.54$) varied significantly along either gradient. This indicates that any effect of these gradients detected on pesticide contamination levels is unlikely to be driven by underlying shifts in species composition or relative abundances.

For each pesticide contamination parameter (i.e., N_C, C_T, $\log_{10}(TI)$), the best-fitting distribution (i.e., Poisson, Negative Binomial, Normal or Gamma distribution) was selected using "fitdistrplus" (Delignette-Muller and Dutang, 2015) and "DHARMa" (Hartig et al., 2024) packages, and based on the lowest Akaike's Information Criterion from the *gofstat* function of the same package. The number of compounds was investigated using models fitted with a Poisson distribution (P-GLMM) for analyses in farmland hedgerows and with a Negative Binomial distribution (NB-GLMM) for analyses in residents' gardens. The total concentration was investigated through models fitted with a Gamma distribution (G-GLMM), whereas linear models (LMM) were used to analyse the $\log_{10}$-transformed toxicity index. The package "glmmTMB" (Brooks et al., 2017) was used to implement the NB-GLMM model and the "lme4" package (Bates et al., 2015) was used for other models.

We investigated how each potential mitigating component (i.e. either the percentage of OF or the distance to the nearest conventional field) affects each parameter of pesticide contamination (i.e., N_C, C_T, $\log_{10}(TI)$). As contamination levels may differ among species, species identity was included as a random intercept to account for baseline differences in pesticide contamination while estimating the overall effect of each mitigating component across the species assemblage. To assess whether species differed in their response to each mitigating component, species-specific slopes were also estimated by including an interaction between the mitigating component and species identity, allowing the relationship between the mitigating component and pesticide contamination to vary among species. Models for the hedgerows dataset therefore included %OF, the date of capture and the interaction between %OF and species identity as fixed factors, with species identity as a random intercept. For residents' gardens, %OF was replaced by the distance from the nearest conventional field, and the municipality (i.e., BSN or FO) was added as a supplementary explanatory variable. The explicit formulation of each model, including the error distribution, link function, fixed effects, interactions, and random-effects structure, is provided in the Supplementary Information (Table S2) to facilitate interpretation and reproducibility. In each environment (i.e., farmland hedgerows and residents' gardens), Moran's I values were close to zero and non-significant ($p > 0.05$) across all models, indicating no evidence of residual spatial autocorrelation and supporting that sites could be treated as independent.

The overall direction of effects and 95% confidence intervals, as well

as species-specific slopes and standard errors (when the interaction between the mitigating component and species identity was significant), were subsequently extracted using the *emtrends* function from the "emmeans" package (Lenth, 2017). For models fitted with a logarithmic link function (Poisson, negative binomial and Gamma models; Table S2), percentage changes in the response variable over a given change in the explanatory variable were calculated as $(\exp(\beta \times \Delta x) - 1) \times 100$, where β is the estimated slope, and Δx is the change in the explanatory variable expressed in the units used in the model. For linear mixed models fitted to $\log_{10}$-transformed toxicity index values using an identity link, the corresponding percentage change on the original toxicity index scale was calculated as $(10^{(\beta \times \Delta x)} - 1) \times 100$. The lower and upper limits of the 95% confidence intervals for β were back-transformed using the same formulas to obtain confidence intervals for the corresponding percentage changes. The statistical significance of each explanatory variable was tested using likelihood-ratio Chi-square tests from the "car" package (Fox and Weisberg, 2019).

2.4.3. Model diagnostics

Model assumptions and goodness-of-fit were assessed for each of the nine GLMM/LMM models (three habitat-comparison models, Section 2.4.1; six mitigating-component models, Section 2.4.2) using the "DHARMa" package (Hartig et al., 2024), based on simulated residuals (1000 simulations per model). For each model, we tested for over-dispersion (*testDispersion*), zero-inflation for count models (*testZeroInflation*), residual uniformity *via* a Kolmogorov-Smirnov test (*testUniformity*), and the presence of influential outliers (*testOutliers*). Model convergence and singularity of the random-effect structure were assessed using the "performance" package (Lüdecke et al., 2021), and marginal R^2 was computed using the "performance" package (Lüdecke et al., 2021; Nakagawa et al., 2017); for models where a near-zero random-effect variance prevented this computation directly, marginal R^2 was instead obtained using the delta method implemented in the "MuMIn" package (Bartoń, 2010). All models showed satisfactory diagnostics and converged properly, with one exception addressed by including a quadratic term for capture date in the total concentration model for the habitat comparison (see Table S3 for details). All nine models nevertheless exhibited a singular fit for the random-effect structure, which is expected given the small number of grouping levels and the presence of fixed species-specific terms. The random-effect structure was retained because it allows species-specific responses to be tested directly. Full diagnostic outputs are provided in Table S3.

Table 3
Summary of the effects of the type of habitat (Hedgerows, Gardens in BSN, Gardens in FO), the species, their interaction, the sex and the date of capture (in Julian days) on the number of compounds, the total concentration and the $\log_{10}$-transformed total toxicity index, analysed respectively using a NB-GLMM, a G-GLMM and a LMM (n=152). *For the G-GLMM of total pesticide concentrations, a quadratic term for capture date was retained. The effects of the site and the species on the number of pesticides, the total concentration and the $\log_{10}$-transformed total toxicity index are shown in Fig. 3. Full regression output (parameter estimates, standard errors, test statistics and p-values for all terms) is provided in Table S4.

Parameters	X^2	*df*	*P*
Number of compounds			
Type of habitat	0.99	2	0.61
Species	3.09	4	0.54
Sex	0.35	1	0.55
Date	0.33	1	0.57
Habitat x Species	7.82	8	0.45
Total concentration			
Type of habitat	0.19	2	0.91
Species	5.58	4	0.23
Sex	1.06	1	0.30
Date*	5.54	2	0.06
Habitat x Species	8.90	8	0.35
$\log_{10}$-transformed total toxicity index			
Type of habitat	0.25	2	0.88
Species	4.55	4	0.34
Sex	0.42	1	0.52
Date	0.26	1	0.61
Habitat x Species	3.02	8	0.93

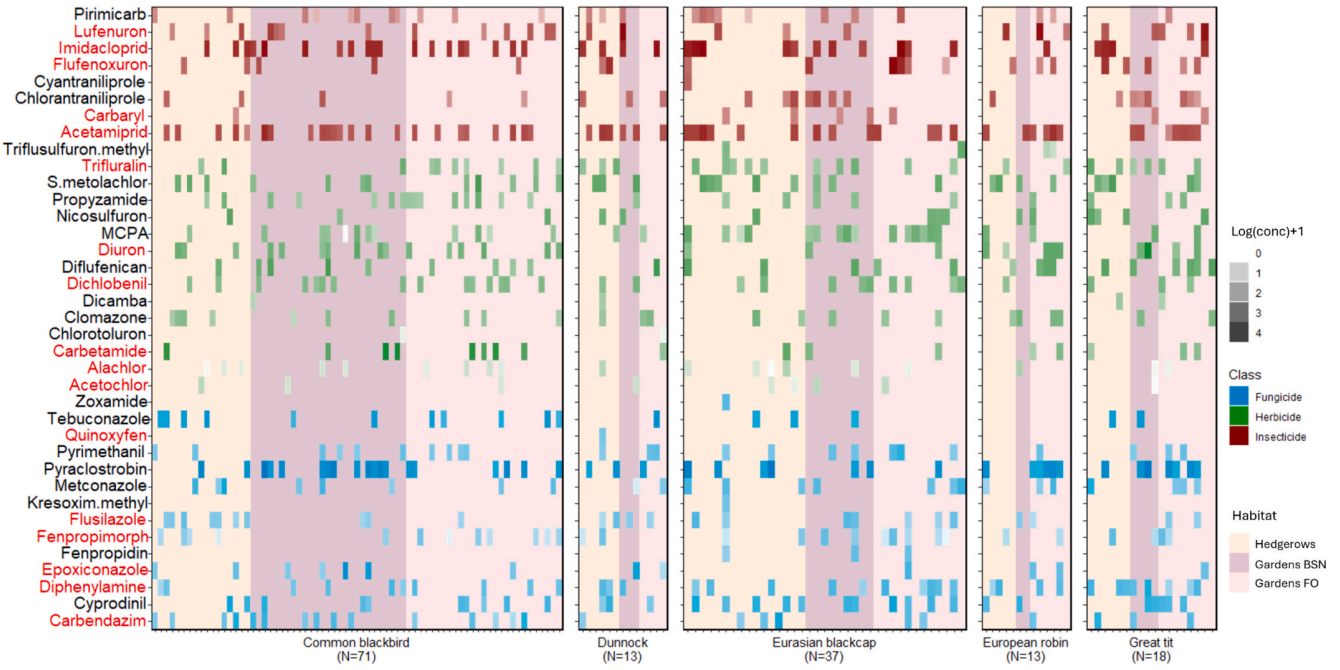


Fig. 2. Heatmap showing the $\log_{10}$-transformed concentrations +1 (in $pg.mg^{-1}$) of the 37 compounds detected in the blood of five passerine species caught in three types of habitats: farmland hedgerows, gardens in BSN, and gardens in FO. Columns represent birds and rows represent pesticide compounds. The compounds are classified according to the pesticide class (fungicides in blue, herbicides in green and insecticides in dark red). The shading indicates the intensity of the concentration: the darker the shading, the higher the concentration. The names of compounds in red are those that were banned in 2022 or earlier.

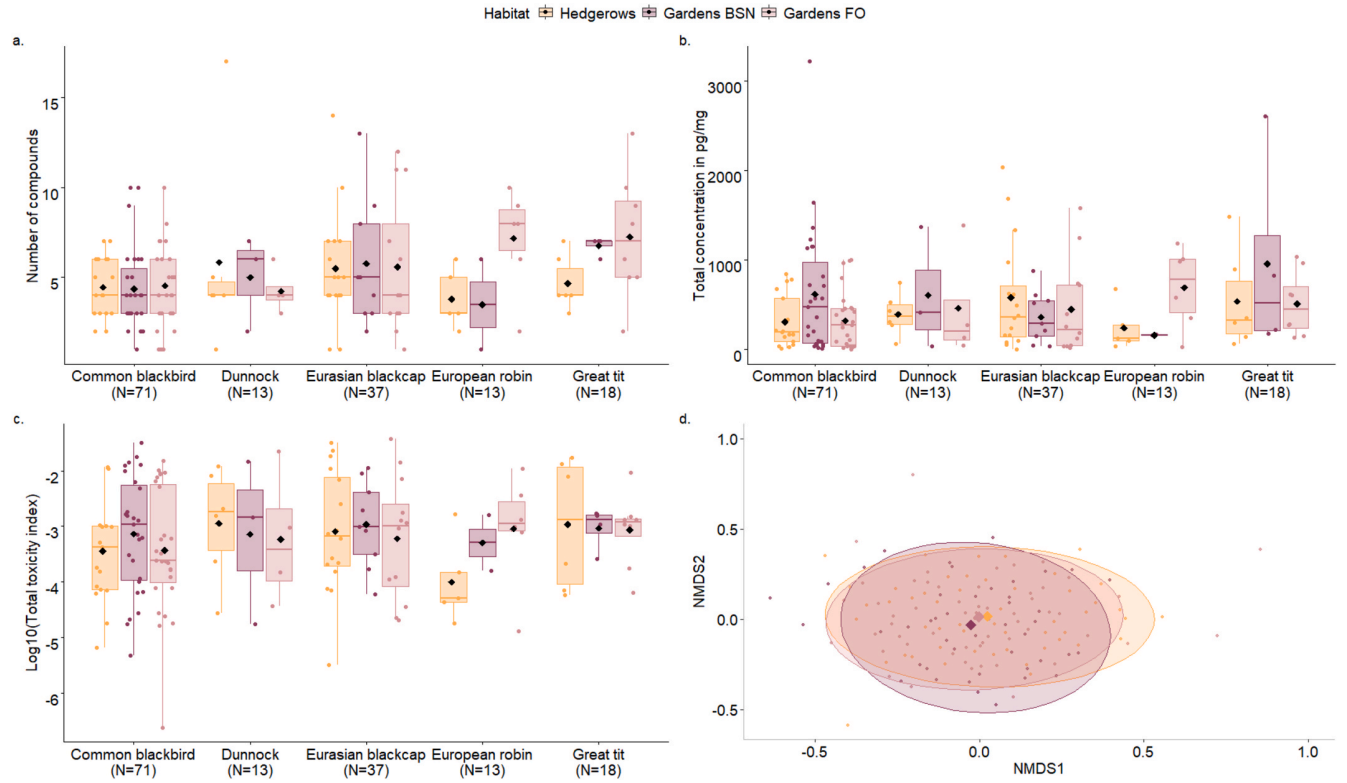


Fig. 3. Effects of the type of habitats (hedgerows, gardens in BSN, gardens in FO) and the species on (a) the number of compounds detected, (b) the total concentration (sum of all compound concentrations in $pg.mg^{-1}$), (c) the log_{10}-transofrmed total toxicity index (sum of all toxicity index) and (d) Jaccard dissimilarities reflecting the composition of pesticide contamination of five passerine species. Each point represents an individual value. In the boxplots, black diamonds represent the mean intensity, the bold line is the median, the box is between the two quartiles (lower edge is the 25th percentile while the upper edge is the 75th percentile), and whiskers extend to 1.5 times the interquartile range. In plot (d), each point represents the pesticide contamination profile of an individual, positioned according to non-metric multidimensional scaling (NMDS) based on Jaccard dissimilarities. Proximity between points reflects similarity in pesticide compound composition. Ellipses represent the dispersion of individuals within each habitat type, and their centroids are symbolised by squares. The sample sizes for each species appear in brackets below each bar, and the sample sizes for each type of farming appear in Table 2.

3. Results

3.1. Comparison of pesticide contamination levels and profiles between habitats

Overall, 37 out of the 111 compounds quantified were found in the five species included for the comparison between farmland hedgerows and gardens (Table 2): 14 fungicides, 15 herbicides, and 8 insecticides (Fig. 2, Table S1). In fact, 17 out of the 37 compounds found, including both farming and domestic uses, have been banned in France before the year of the study (Fig. 2). Eurasian Blackcaps had the highest compound diversity, with 34 of the 37 compounds detected; Common Blackbirds had 31, Great Tits and Dunnocks had 30, and European Robins had 27 compounds detected. Among the 37 compounds detected, 62% were strictly used in farming areas (23 compounds out of 37 compounds detected), 35% were for farming and domestic use (13 compounds) and

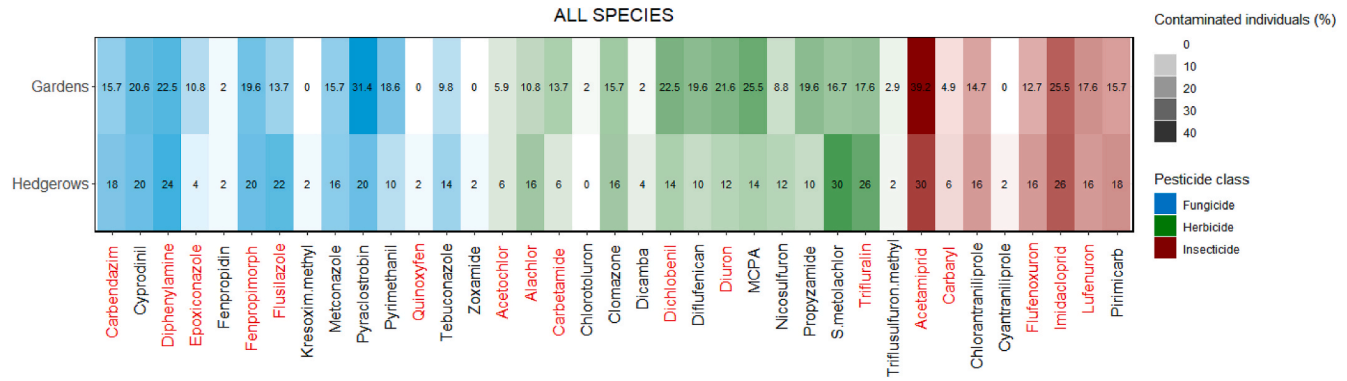


Fig. 4. Heatmap showing the comparison of the percentage of birds (from five passerine species) contaminated by each of the 37 compounds detected (in columns) between two habitats (in rows): farmland hedgerows and gardens (including both villages, i.e., BSN and FO). The compounds are classified according to the pesticide class (fungicides in blue, herbicides in green and insecticides in dark red). The shading indicates the intensity of the prevalence of contamination: the darker the shading, the higher the proportion of contaminated individuals. The names of compounds in red are those that were banned in 2022 or earlier.

3% were for farming, domestic and veterinary use (one compound, imidacloprid, banned for farming use since 2018).

The average number of compounds detected per bird was 4.92 ± 2.85 in farmland hedgerows, 4.87 ± 2.62 in BSN gardens, and 5.40 ± 2.97 in FO gardens (Tables 2 and 3, Fig. 3a). Pairwise comparisons provided no clear evidence of differences among habitats, with all 95% confidence intervals for the ratios of estimated marginal means including 1 (hedgerows vs. BSN gardens: ratio = 0.97, 95% CI [0.69; 1.37]; hedgerows vs. FO gardens: ratio = 0.86, 95% CI [0.65; 1.13]; BSN vs. FO gardens: ratio = 0.88, 95% CI [0.63; 1.24]). Similarly, average total pesticide concentration was 429.57 ± 452.00 pg.mg^{-1} in farmland hedgerows, 579.25 ± 675.04 pg.mg^{-1} in BSN gardens, and 425.99 ± 409.43 pg.mg^{-1} in FO gardens (Tables 2 and 3, Fig. 3b). Pairwise contrasts again provided no clear evidence of differences among habitats (hedgerows vs. BSN gardens: ratio = 0.88, 95% CI [0.46; 1.65]; hedgerows vs. FO gardens: ratio = 0.85, 95% CI [0.50; 1.45]; BSN vs. FO gardens: ratio = 0.97, 95% CI [0.51; 1.84]). The $\log_{10}$-transformed toxicity index showed a similar pattern, with all pairwise confidence intervals including zero (hedgerows vs. BSN gardens: difference = −0.17, 95% CI [−0.87; 0.53]; hedgerows vs. FO gardens: difference = −0.07, 95% CI [−0.71; 0.56]; BSN vs. FO gardens: difference = 0.09, 95% CI [−0.64; 0.83]; Tables 2 and 3, Fig. 3c). No statistically significant effects of species, sex, or capture date were detected for the three contamination parameters (Table 3). Pesticide contamination profiles also showed no statistically significant differences among habitats (PERMANOVA based on Jaccard dissimilarities, p = 0.20; Fig. 3d), among species (p = 0.53), or for the habitat × species interaction (p = 0.52).

Indeed, all 37 compounds were detected in all types of habitats (hedgerows, both village gardens, Fig. 4) except for one, found only in gardens, chlorotoluron (herbicide for farming use) though only in two birds, and for four other compounds (kresoxim methyl, zoxamide, quinoxyfen (fungicides) and cyantraniliprole (insecticide), all used for crops), which were only detected in birds caught in hedgerows (only in one individual each time). The most common compound found was acetamiprid, an insecticide from the neonicotinoid family, detected in all bird species and in 39% and 30% of birds in gardens and hedgerows, respectively (Fig. 4), despite its ban from use in France since 2018.

Table 4
Summary of the effects of the percentage of organic farming (%OF) in farmland hedgerows, the date of capture (in Julian days) and the interaction between the percentage of organic farming and species, with the species also added as a random variable on the number of compounds, the total concentration and $\log_{10}$-transformed total toxicity index, analysed respectively using a P-GLMM, a G-GLMM and a LMM (n=67). The effects of the percentage of organic farming on the number of pesticides, the total concentration and the total toxicity index for all species are shown in Fig. 5. The effects of the percentage of organic farming for each species on the number of pesticides, the total concentration and the total toxicity index are shown in Fig. S1, S2 and S3 respectively. Full regression output (parameter estimates, standard errors, test statistics and p-values for all terms) is provided in Table S5.

Parameters	X^2	*Df*	*P*
Number of compounds			
%OF	0.05	1	0.82
Date	**6.74**	**1**	**<0.01**
%OF : Species	9.16	6	0.16
Total concentration			
%OF	**9.03**	**1**	**<0.01**
Date	0.50	1	0.48
%OF : Species	**20.38**	**6**	**<0.01**
$\log_{10}$-transformed total toxicity index			
%OF	0.28	1	0.60
Date	0.17	1	0.68
%OF : Species	7.28	6	0.30

3.2. Effect of organic farming on pesticide contamination levels in farmland hedgerows

Considering all seven species, the number of compounds and the $\log_{10}$-transformed toxicity index did not differ significantly according to the percentage of organic farming in a 250m-radius buffer (Table 4; N_C: β = 0.0005, 95%CI [−0.0032; 0.0041], Fig. 5a; $\log_{10}$(TI): β = −0.0018, 95%CI [−0.0106; 0.0069], Fig. 5c). However, total pesticide concentration in hedgerows decreased significantly with increasing %OF (Table 4, Fig. 5b), with an estimated global decrease of 61.3% (95% CI [32.5; 77.8]) across the full gradient of organic farming (0–100%) (β = −0.0095, 95%CI [−0.0151; −0.0039]). Model-predicted total pesticide concentrations decreased from 643.99 pg.mg^{-1} (95% CI [460.27; 901.04]) at 0% OF to 400.54 pg.mg^{-1} (95% CI [326.13; 491.93]) at 50% OF and 249.13 pg.mg^{-1} (95% CI [174.49; 355.69]) at 100% OF (Table S7).

We found a significant interaction between the percentage of organic farming (%OF) and species identity for the total concentration (Table 4), indicating that the effect of %OF differed among species. Total concentration decreased significantly with increasing %OF for the Common Blackbird, European Robin, Great Tit, and Melodious Warbler (Fig. 7, Fig. S2). In contrast, the interaction between %OF and species identity was not significant for the number of compounds and the $\log_{10}$(TI) (Table 4). Nevertheless, species-specific trends were predominantly negative, with only two positive associations detected for the number of compounds (Eurasian Blackcaps and Melodious Warblers). Only the European Robin showed consistent negative responses across all three contamination parameters, with moderate decreases also observed for the number of compounds and the $\log_{10}$(TI) (Fig. 7, Figs. S1 and S3). In contrast, none of the three pesticide contamination parameters were influenced by %OF in the Common Nightingale (Fig. 7).

Moreover, the date of capture did not influence the total concentration and the $\log_{10}$(TI), whereas the number of compounds decreased significantly with Julian date (Table 4), corresponding to an estimated decrease of 41.2% (95% CI [12.2; 60.6]) across the range of capture dates (from 4th April to 17th July).

3.3. Effect of the distance from the nearest conventional field on pesticide contamination levels in residents' gardens

Considering all nine species, all three pesticide contamination parameters decreased significantly with increasing distance from the nearest conventional field (Table 5), with estimated global decreases of 30.1% (95% CI [6.8; 47.6]) for the number of compounds (Fig. 6a), 70.1% (95% CI [49.2; 82.4]) for total concentration (Fig. 6b), and 85.0% (95% CI [41.0; 96.2]) for the toxicity index (Fig. 6c) across the full gradient of distance (0–378m) (N_C: β = −0.0009, 95%CI [−0.0017; −0.0002]; C_T: β = −0.0032, 95%CI [−0.0046; −0.0018]; $\log_{10}$(TI): β = −0.0022, 95%CI [−0.0037; −0.0006]). For instance, model-predicted total pesticide concentrations decreased from 713.08 pg.mg^{-1} (95% CI [556.34; 913.98]) at 0 m to 376.65 pg.mg^{-1} (95% CI [318.38; 445.58]) at 200 m and 273.74 pg.mg^{-1} (95% CI [210.01; 356.80]) at 300 m from the nearest conventional field (Table S8).

Similarly to %OF, the interaction between the distance from the nearest conventional field and species identity was significant only for total concentration (Table 5). Total concentration decreased significantly with increasing distance for the Common Blackbird, Eurasian Blackcap, European Goldfinch, and European Robin (Fig. 7, Fig. S5). Although the interaction was not significant for the number of compounds or $\log_{10}$(TI), the number of compounds nevertheless decreased significantly with increasing distance in the Black Redstart and Common Blackbird (Fig. 7, Fig. S4). Only the Common Blackbird and the European Robin showed consistent negative responses across all three contamination parameters, with moderate decreases also observed for the $\log_{10}$(TI) (Fig. 7, Fig. S6). Other species-specific trends were predominantly negative, even not significant, as all three contamination

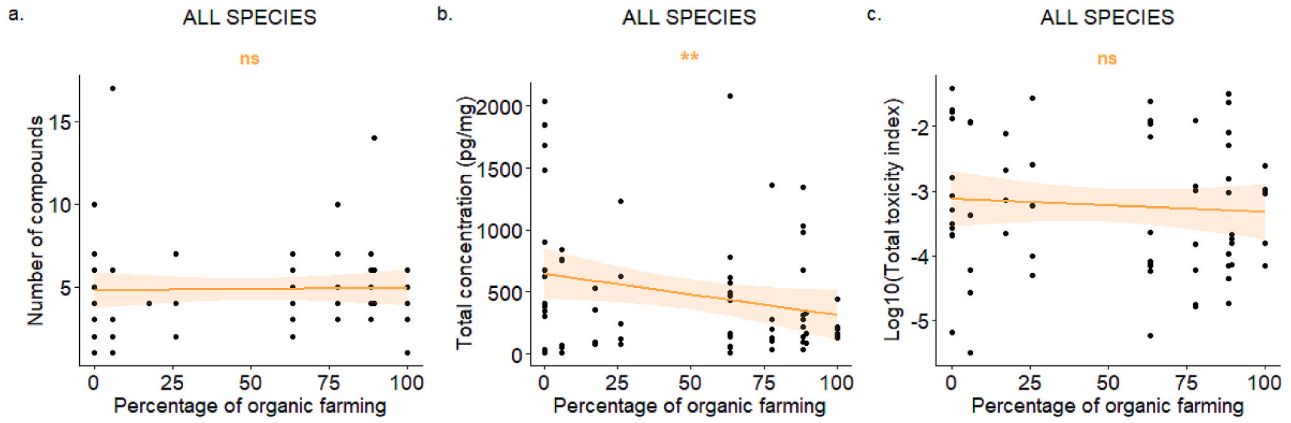


Fig. 5. Global effects of the percentage of organic farming for all species caught in farmland hedgerows and included in the analyses on (a) the number of compounds detected, (b) the total concentration (sum of all compound concentrations in $pg.mg^{-1}$) and (c) the log_{10}-transformed total toxicity index (sum of all toxicity index), analysed respectively with a NB-GLMM, a G-GLMM and a LMM. In these models, %OF, date of capture and the interaction between %OF and the species were included as fixed factors, and the species was also added as a random effect. The line and the shading represent the predicted values and 95% confidence interval from a simple linear regression fitted across all species pooled together. For true estimate, please report to Table S4. The results of the statistical effect of the percentage of organic farming on each pesticide contamination parameter are indicated with "ns." for non-significant; "*" for p-values < 0.05; "**" for p-values < 0.01; "***" for p-values < 0.001. The effects of the percentage of organic farming for each species are shown in Figs. S1, S2 and S3 for the number of compounds detected, total concentrations and total toxicity index, respectively.

Table 5
Summary of the effects of the distance from the nearest conventional crop in gardens, the date of capture (in Julian days), the municipality (BSN and FO) and the interaction between the distance from the nearest conventional crop and species, with the species also added as a random variable on the number of compounds, the total concentration and the log_{10}-transformed total toxicity index, analysed respectively using a NB-GLMM, a G-GLMM and a LMM (n=142). The effects of the distance from the nearest conventional crop on the number of pesticides, the total concentration and the total toxicity index for all species are shown in Fig. 6. These effects for each species on the number of pesticides, the total concentration and the total toxicity index are shown in Fig. S4, S5 and S6 respectively. Full regression output (parameter estimates, standard errors, test statistics and p-values for all terms) is provided in Table S6.

Parameters	X^2	*Df*	*P*
Number of compounds			
Distance	**9.33**	**1**	**0.002**
Municipality	1.00	1	0.32
Date	1.26	1	0.26
Distance : Sp ecies	5.95	8	0.65
Total concentration			
Distance	**17.17**	**1**	**<0.001**
Municipality	0.89	1	0.35
Date	0.19	1	0.66
Distance : Specie s	**20.19**	**8**	**<0.01**
Log_{10}-transformed total toxicity index			
Distance	**6.35**	**1**	**0.01**
Municipality	0.00	1	0.97
Date	0.73	1	0.39
Distance : Spec ies	6.36	8	0.61

parameters decreased with increasing distance for the Dunnock, and two contamination parameters decreased for the Common Linnet (total concentration and log_{10}(TI)). Only the total concentration decreased with the increasing distance from the nearest conventional field for the Eurasian Chaffinch and none of the three contamination parameters was influenced by the distance for the Great Tit. None of the other variables (date of capture and municipalities, i.e., BSN and FO) significantly affected any of the three contamination parameters.

4. Discussion

This study aimed to compare pesticide contamination in passerine birds caught in farmland hedgerows and residents' gardens and to explore whether organic farming and the distance from conventional fields could reduce contamination levels in both environments. We found widespread pesticide contamination across habitats (i.e., farmland hedgerows, gardens in both municipalities), with similar contamination levels and profiles. Despite the ubiquitous contamination generalized to all environments, our results also highlighted a reducing effect of organic farming only on pesticide concentrations in hedgerows. In addition, all three parameters in birds caught in gardens (i.e., number of compounds, total concentration and toxicity index) decreased significantly with the increase in the distance to the nearest conventional field. Although these associations are consistent with a mitigating effect of both factors, they still need to be confirmed and validated with larger sample sizes and a broader range of species. Passerine birds may nonetheless be valuable bioindicators of the pesticide pressure applied in the environment and of the potential positive effects of both organic farming and distance from treated crops.

4.1. *Widespread pesticide contamination of passerines in agricultural environments and residents' gardens*

Among the 111 pesticide compounds screened, 37 were detected in the five passerine species commonly found in both farmland hedgerows and gardens, reflecting the broad diversity of molecules already reported in farmland birds in our study area (Bailly et al., 2025; Fuentes et al., 2023), although direct comparisons remain limited by differences in sampling sites and species across studies. Consistent with previous findings, our study confirms the presence of some compounds in passerines years after their ban, suggesting that some compounds may persist in the environment longer than estimated. Despite bans on certain compounds, many have been commercially available for decades to both farmers and non-farmer residents. The fraudulent or inadvertent use of such leftover products, which remain in circulation long after their official withdrawal from the market, cannot be ruled out (Bailly et al., 2025). Moreover, regulatory bans are limited to national or European levels, while many of these compounds remain authorised and in use elsewhere in the world. Their continued global use may therefore contribute to environmental persistence and long-range atmospheric or trophic transport, independently of illegal or fraudulent use within national borders.

Surprisingly, we detected no significant differences in pesticide contamination levels (i.e., number of compounds, total concentration,

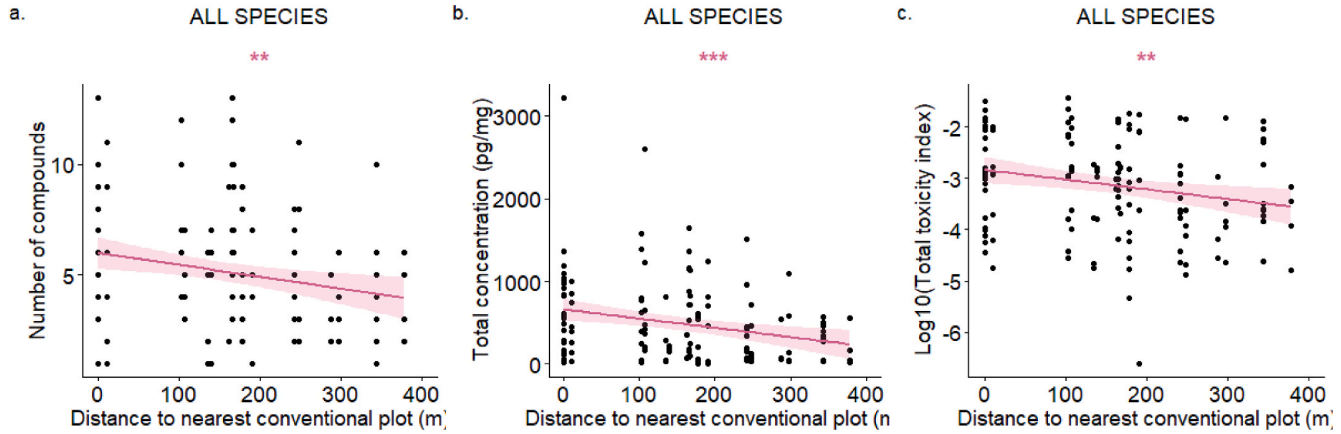


Fig. 6. Global effects of the distance from the nearest conventional crop for all species caught in gardens of the two municipalities and included in the analyses on (a) the number of compounds detected, (b) the total concentration (sum of all compound concentrations in $pg.mg^{-1}$) and (c) the log_{10}-transformed total toxicity index (sum of all toxicity index), analysed respectively with a NB-GLMM, a G-GLMM and a LMM. In these models, the distance from the nearest conventional crop, date of capture, the municipality and the interaction between the distance from the nearest conventional crop and the species were included as fixed factors, and the species was also added as a random effect. The line and the shading represent the predicted values and 95% confidence interval from a simple linear regression fitted across all species pooled together. For true estimate, please report to Table S5. The results of the statistical effect of the distance from the nearest conventional crop on each pesticide contamination parameter are indicated with "ns." for non-significant; "*" for p-values < 0.05; "**" for p-values < 0.01; "***" for p-values < 0.001. The effects of distance from the nearest conventional crop for each species are shown in Figs. S4, S5 and S6 for the number of compounds detected, total concentrations and total toxicity index, respectively.

and toxicity index) or contamination profiles (i.e., presence of specific compounds) between birds captured in agricultural hedgerows and those from residential gardens. Contrary to expectations, birds in non-agricultural areas such as gardens were not less contaminated than those caught in farmland. Agricultural areas are subject to intensive pesticide use, sometimes at high levels. For example, in 2021, winter wheat crops in the Nouvelle-Aquitaine region received an average of 3.6 pesticide treatments (SSP and Agreste, 2024). As a result, non-target species are highly contaminated by pesticides in agricultural environments, as reported in our study area across multiple taxa, including earthworms (Pelosi et al., 2021, 2022), small mammals (Fritsch et al., 2022), passerine birds (Bailly et al., 2025), Grey Partridges (Bariod et al., 2025) and Montagu's Harriers (Fuentes et al., 2023, 2025). However, in urban environments, such as in residents' gardens, far fewer pesticides are used. A reduction in pesticide contamination in birds from gardens compared to those from farmland was therefore expected, but not observed. Moreover, the European Commission's proposal for a Regulation on the Sustainable Use of Pesticides (SUR) calls for 3-metre buffer zones with no pesticide use between sprayed fields and inhabited buildings to reduce pesticide exposure for residents living near treated fields. In France, based on the recommendations of the French Agency for Food, Environmental and Occupational Health & Safety, larger buffer zones were established since 1st of January 2020 (5, 10 or 20 metres according to the type of crop, Ministère de l'agriculture et de la souveraineté alimentaire, 2025). Such non-treatment buffer zones were expected to reduce contamination in residential areas. Because our results showed that pesticide contamination in birds from gardens remained comparable to that in birds from agricultural hedgerows, the effectiveness of these measures is questionable. This suggests that current buffer distances may be insufficient to prevent pesticide dispersal beyond treated areas, possibly due to rainwater run-off (Chen et al., 2019) or spray drift during or shortly after application (Albaseer et al., 2025; Felsot et al., 2010; Hong et al., 2021; Lazzaro et al., 2008; Zhang et al., 2024). Many studies have demonstrated that a substantial proportion of pesticides drift away from target areas during application (Albaseer et al., 2025; Hong et al., 2021). The extent of aerial drift can depend on the type of crop and spraying technique. For most crops, drift fractions due to pesticide application are less than 3% of the mass applied, whereas higher values are observed in vineyards and orchards (up to ~5%), and extreme cases occur with aerial spraying, where drift can reach 20–60% of the pesticide mass applied to soybean fields, for instance (Zhang et al., 2024). Other studies have reported that between 10 and 60% of applied pesticides can travel more than 300 m from the treated fields, while over 70% of deposits were detected as far as 195m downwind (Salyani and Cromwell, 1992; van den Berg et al., 1999). Such pesticide dispersal may also explain why the increase in organic farming did not affect the number of compounds detected or the toxicity index in birds caught in farmland hedgerows. Indeed, birds caught in organic environments had surprisingly the same compound diversity as those caught in strictly conventional environments, underscoring ubiquitous contamination across environments, including organic farming, despite zero synthetic pesticide pressure. Moreover, since the expansion of organic farming is relatively recent, birds caught in organic farming may still be indirectly exposed (e.g., through contaminated food or water, soil, air) to residual compounds applied before conversion. Given the long environmental persistence of certain pesticides, further analyses that consider the year of conversion to organic farming are needed to better understand pesticide contamination in environments where synthetic pesticide use has ceased.

High levels of pesticide contamination in villages may also be explained by residents applying pesticides in their gardens, vegetable gardens, or homes, although this is unlikely, as we did not find habitat-specific contamination profiles. Nonetheless, pesticide use in urban and domestic settings is widespread and not limited to agriculture, with applications occurring in home gardens, lawns, and other non-agricultural areas, in significant quantities (Md Meftaul et al., 2020; Rani et al., 2021; Tessier et al., 2023). For instance, a study conducted in the United Kingdom between 2020 and 2021 reported that a substantial proportion of residents (32.2% of surveyed gardens) apply pesticides in their gardens, most commonly glyphosate, and that such practices can have detrimental effects on local wildlife (i.e., reductions in bird abundance and species richness, Tassin de Montaigu and Goulson, 2023). Another study conducted in France in 2014 (Pesti'home survey) reported even higher usage rates, with 75% of households indicating that they had used at least one pesticide in the previous 12 months (Tessier et al., 2023). It is also estimated that 80% of dogs and 82% of cats are treated for fleas at least once a year using an insecticide, and significant quantities of fipronil, imidacloprid and permethrin (used

Bird species	Parameters	Effect of %OF in hedgerows	β ± Std. Error P-value	Effect of dist. in gardens	β ± Std. Error P-value
Black Redstart 	N_C	-	-	↘	**-0,0012 ± 0,0006 P = 0,04**
	C_T	-	-	↘	-0,0012 ± 0,0010 P = 0,21
	$Log_{10}(TI)$	-	-	→	-0,0004 ± 0,0015 P = 0,79
Common Blackbird 	N_C	→	-0,0011 ± 0,0019 P = 0,55	↘	**-0,0012 ± 0,0004 P < 0,01**
	C_T	↘	**-0,0094 ± 0,0035 P < 0,01**	↘	**-0,0028 ± 0,0007 P < 0,001**
	$Log_{10}(TI)$	→	-0,0032 ± 0,0074 P = 0,70	↘	-0,0021 ± 0,0015 P = 0,31
Common Linnet 	N_C	-	-	→	-0,0004 ± 0,0010 P = 0,71
	C_T	-	-	↘	-0,0031 ± 0,0019 P = 0,11
	$Log_{10}(TI)$	-	-	↘	-0,0029 ± 0,0023 P = 0,23
Common Nightingale 	N_C	→	-0,0022 ± 0,0027 P = 0,42	-	-
	C_T	→	0,0025 ± 0,0046 P = 0,59	-	-
	$Log_{10}(TI)$	→	0,0020 ± 0,0072 P = 0,80	-	-
Dunnock 	N_C	→	-0,0012 ± 0,0035 P = 0,74	↘	-0,0025 ± 0,0015 P = 0,09
	C_T	↘	-0,0084 ± 0,0063 P = 0,18	↘	-0,0030 ± 0,0022 P = 0,18
	$Log_{10}(TI)$	→	0,0047 ± 0,0099 P = 0,64	↘	-0,0039 ± 0,0027 P = 0,16
Eurasian Blackcap 	N_C	↗	0,0032 ± 0,0022 P = 0,16	→	-0,0004 ± 0,0007 P = 0,54
	C_T	→	-0,0028 ± 0,0045 P = 0,53	↘	**-0,0032 ± 0,0013 P = 0,02**
	$Log_{10}(TI)$	→	0,0035 ± 0,0086 P = 0,70	→	-0,0014 ± 0,0021 P = 0,58
Eurasian Chaffinch 	N_C	-	-	→	-0,0009 ± 0,0011 P = 0,41
	C_T	-	-	↘	-0,0023 ± 0,0021 P = 0,26
	$Log_{10}(TI)$	-	-	→	-0,0015 ± 0,0027 P = 0,59
European Goldfinch 	N_C	-	-	→	-0,0011 ± 0,0011 P = 0,35
	C_T	-	-	↘	**-0,0087 ± 0,0020 P < 0,0001**
	$Log_{10}(TI)$	-	-	↘	-0,0042 ± 0,0025 P = 0,11
European Robin 	N_C	↘	-0,0049 ± 0,0040 P = 0,22	↘	-0,0020 ± 0,0015 P = 0,20
	C_T	↘	**-0,0198 ± 0,0062 P < 0,01**	↘	**-0,0093 ± 0,0025 P < 0,001**
	$Log_{10}(TI)$	↘	-0,0153 ± 0,0092 P = 0,14	↘	-0,0052 ± 0,0032 P = 0,12
Great Tit 	N_C	→	-0,0044 ± 0,0060 P = 0,47	→	0,0010 ± 0,0011 P = 0,41
	C_T	↘	**-0,0299 ± 0,0096 P < 0,01**	→	-0,0014 ± 0,0023 P = 0,56
	$Log_{10}(TI)$	↘	-0,0146 ± 0,0147 P = 0,34	→	-0,0026 ± 0,0032 P = 0,4
Melodious Warbler 	N_C	↗	0,0036 ± 0,0024 P = 0,13	-	-
	C_T	↘	**-0,0135 ± 0,0047 P < 0,01**	-	-
	$Log_{10}(TI)$	→	-0,0035 ± 0,0081 P = 0,69	-	-

Fig. 7. Summary of the effects of the percentage of organic farming (%OF) in hedgerows and the effects of the distance from the nearest conventional crop in gardens for each species and all pesticide contamination parameters (i.e., number of compounds N_C, total concentrations C_T and total toxicity index TI). For each species and parameter, arrows indicate the direction of the relationship (up: positive; down: negative; flat: no discernible effect), and their colour reflects the effect's statistical strength: dark orange (effect of %OF in hedgerows) or dark purple (effects of the distance from the nearest conventional crop in gardens) arrows indicate significant effects ($P < 0.05$); light orange or light purple arrows indicate non-significant effects for which the estimate ± standard error did not overlap zero; grey flat arrows indicate non-significant effects for which the estimate ± standard error overlapped zero. The effects of the percentage of organic farming for each species are shown in Figs. S1, S2 and S3 for the number of compounds, total concentrations and total toxicity index, respectively. The effects of the distance from the nearest conventional crop for each species are shown in Figs. S4, S5 and S6 for the number of compounds, total concentrations and total toxicity index, respectively.

as insecticides in domestic pets) were detected in the nests of blue tits and great tits (Tassin de Montaigu et al., 2025). Future analysis of the surveys conducted among the residents of our study (including a list of compounds applied, application doses, number of applications, and application dates) could better explain contamination sources and their links to agricultural and domestic uses.

4.2. Mitigating components of pesticide contamination levels

Both organic farming and distance from the nearest conventional crop seem to be relevant factors in mitigating pesticide contamination levels. Indeed, distance from the nearest conventional crop significantly influenced pesticide contamination across all nine species caught in residents' gardens, as the number of compounds, total concentration, and toxicity index decreased by 30.1%, 70.1%, and 85.0%, respectively. Although we cannot exclude the possibility that residents themselves apply pesticides to their gardens, as previously described, our findings therefore indicate that the main source of contamination remains pesticides applied to conventional crops. These results are consistent with our hypothesis that contamination in birds may decrease with distance from the main source of contamination, as is the case in other matrices such as household dust and air (Dereumeaux et al., 2020; Deziel et al., 2017; Gibbs et al., 2017; Gunier et al., 2011; Hogenkamp et al., 2004; Kawahara et al., 2005; Teysseire et al., 2021; Ward et al., 2006; Wofford et al., 2014) and residents' biological samples (Chevrier et al., 2014; Mercadante et al., 2013; Van Wendel De Joode et al., 2012; Wu et al., 2013). Although pesticide dispersal can extend contamination beyond treated crops, exposure generally remains highest at field edges, within a few meters to tens of meters, as AgDrift models predict (Teske et al., 2002). Some more volatile compounds may travel longer distances, but their concentrations can be progressively diluted, resulting in lower contamination levels several hundred meters from treated areas. This dilution can be provided by landscape elements such as hedgerows and shrubs, as they intercept and concentrate pesticide drift (Kjær et al., 2014; Lazzaro et al., 2008; Otto et al., 2009). For instance, off-site spray reductions between 82 and 97% have been reported at the edge of fields containing at least one hedgerow (Lazzaro et al., 2008). Nevertheless, the extent of pesticide drift is strongly influenced by meteorological conditions (e.g., wind currents), local topography (e.g., relief that can shape water run-off), and landscape structure (Moore et al., 2021). Thus, even though contamination typically decreases with increasing distance from application sites, further analyses integrating these environmental variables together with pesticide application practices (e.g., type of sprayer, droplet size, aerial vs. ground spraying) are required to better understand the links between pesticide application and exposure in residents' gardens. Notably, reduced contamination levels were observed over long distances, as the most distant sites are located 378 m from treated crops, raising questions about the actual effectiveness of the regulatory distances established (5, 10 or 20 metres according to the type of crop, Ministère de l'agriculture et de la souveraineté alimentaire, 2025). These results therefore suggest that the current buffer zones may be insufficient to limit pesticide exposure for organisms living near agricultural areas, as it was already proposed in Linhart et al. (2021) for instance.

In addition, although organic farming did not allow to mitigate the number of compounds detected or the total toxicity index, the overall total concentration (including seven species) decreased by ~61.3% when the percentage of organic farming within a 250 metres radius increased (from 696.96 $pg.mg^{-1}$ on average in sites with less than 10% of OF to 221.06 $pg.mg^{-1}$ on average in sites with more than 90% of OF). A reduction in pesticide concentrations in birds caught in organic farming has already been reported in other studies (on Montagu's harrier chicks, Fuentes et al., 2024a), although this effect may have been species-dependent (a decrease of 55% and 90% of total concentrations in OF for Common Blackbirds and Eurasian Blackcaps, respectively, Bailly et al., 2025). A first explanation for these lower concentrations as the organic farming gradient increases is that organic farming prohibits the use of synthetic pesticides (Van Mansvelt et al., 1998). Pesticide concentrations in the environment may be reduced, and birds may therefore be less contaminated by pesticides due to lower exposure through inhalation (Mineau, 2011; Sánchez-Bayo, 2011), dermal absorption (Driver et al., 1991; Hudson et al., 1979) or ingestion of contaminated food (e.g., seeds, insects) and water (Lopez-Antia et al., 2016; Mineau, 2011). Another, but not mutually exclusive, explanation is that birds foraging in environments mainly in organic farming generally have access to a more diverse food supply, resulting in a higher-quality diet (Blackburn and Wallace, 2001; Girard et al., 2014). Such diets can be richer in carotenoids, which are known to play an important role in physiological detoxification processes. Consequently, birds feeding on carotenoid-rich diets may better mitigate contaminant-induced physiological stress, ultimately contributing to lower pesticide loads (Møller et al., 2000).

4.3. Passerine birds as bioindicators of the variability of pesticide pressure

Both the proportion of organic farming and distance from the nearest conventional crop significantly affected pesticide contamination, especially total pesticide contamination. However, our results showed contrasting effects of the mitigating components on pesticide contamination across species (Fig. 7). For instance, pesticide contamination was negatively associated with the percentage of organic farming for Great Tits and European Robins caught in farmland hedgerows, whereas no association was found for Common Nightingales. Birds caught in gardens also reflect this high variation between species: pesticide contamination in Black Redstarts, Common Blackbirds, Eurasian Blackcap, European Goldfinches and European Robins was negatively associated with the distance to the nearest conventional crop, whereas there was no association detected for Great T its for example. These species-dependent responses prevent us from generalising our findings to all bird populations and could be explained by several factors. It could result from physiological differences in the elimination of contaminants. Indeed, metabolic pathways and biotransformation rates can vary depending on both pesticide compounds (Katagi and Fujisawa, 2021; Li, 2023), and bird species through variations in body size, metabolic enzyme activity and feeding habits (Cooper et al., 2017; Kuo et al., 2022; Lucia et al., 2012; Rainio et al., 2012).

Our main hypothesis is linked to the contrasting ecologies of the species (Fig. 8), especially regarding their diet (i.e., insectivorous, granivorous, frugivorous, mixed diet), habitat use (e.g., urban or agricultural specialists, or spending a lot of time on the ground, in woodlands, in hedgerows), migratory status (i.e., migratory, partial migratory or sedentary) and home range size during the breeding season (Gillings et al., 2026). Although passerine species in our study are generally not highly mobile during the breeding season, their home ranges range from 0.2 to 5 hectares (corresponding to circular buffers with a radius between 20 and 130 metres). Consequently, we can assume that species with very restricted home ranges could better represent local contamination and, therefore, local pesticide pressure. For example, Common Blackbirds have a larger home range in farmlands and forests (estimated at 4.7 hectares for Common Blackbirds, Ferry et al., 1981) than European Robins (estimated at 0.84 ha in Kirwan et al., 2025, or at 2.1 ha in Ferry et al., 1981) and Great Tits (estimated at 1.2 ha on average, Wilkin et al., 2009, Fig. 8). Despite limited sample sizes, these last two species showed the strongest responses to the reduction in pesticide pressure in farmland hedgerows (where contamination decreased with increasing % OF).

In contrast, Common Blackbirds showed limited responses in farmland hedgerows, consistent with their comparatively larger home range. Common Blackbird home ranges are known to shrink considerably in urban environments (estimated between 0.16 and 0.24 ha, Wysocki et al., 2004), likely reflecting higher food resource and habitat availability in gardens. This reduced home range size in urban settings may

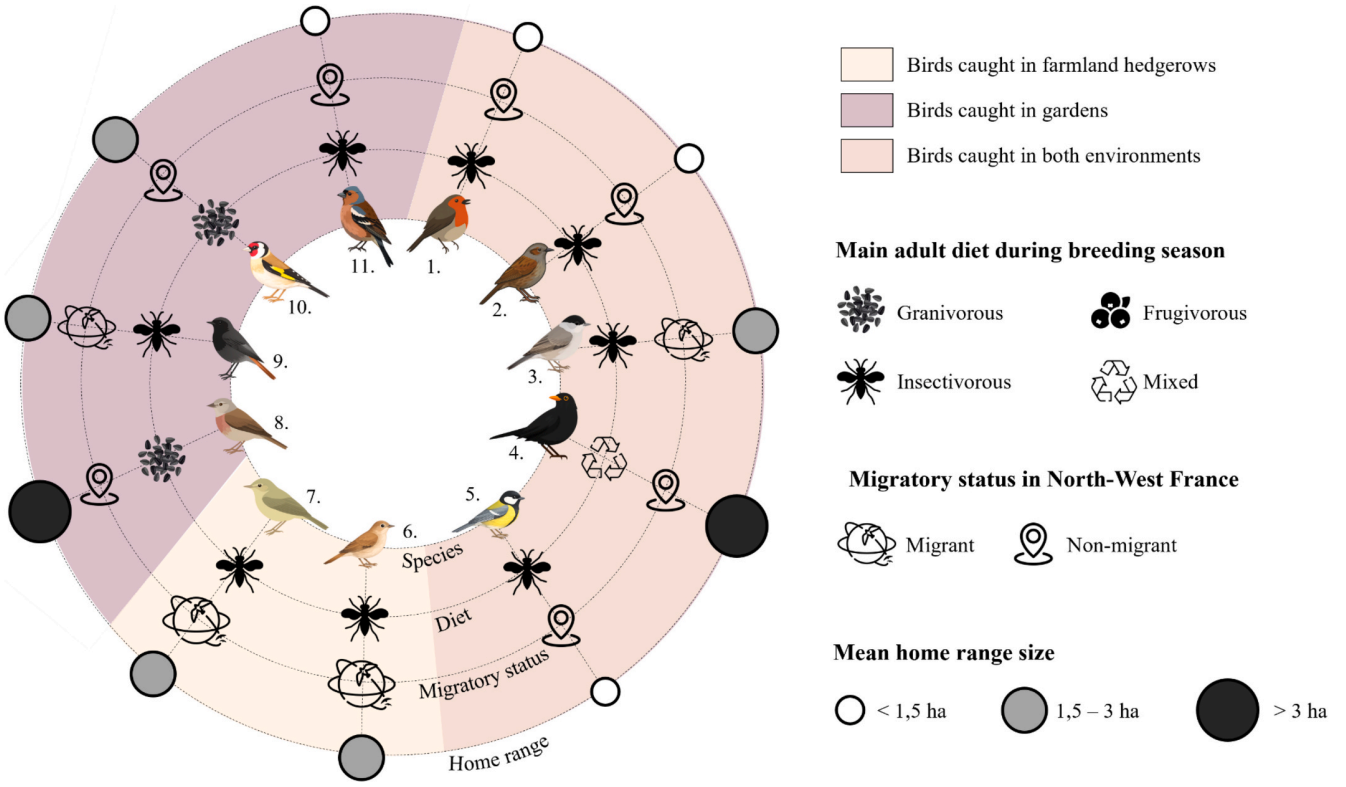


Fig. 8. Ecological parameters, i.e., the main adult diet during breeding season, the migratory status in North-West France and the mean home size, for the eleven species caught in farmland hedgerows and gardens. The species caught in both environments are 1. European Robin, 2. Dunnock, 3. Eurasian Blackcap, 4. Common Blackbird, 5. Great Tit. Those caught only in farmland hedgerows are 6. Common Nightingale and 7. Melodious Warbler. Those caught only in gardens are 8. Common Linnet, 9. Black Redstart, 10. European Goldfinch, 11. European Chaffinch.

explain why, in contrast to hedgerows, Common Blackbirds showed a clear negative association between pesticide contamination and distance from the nearest conventional field in gardens: a smaller home range in this context likely allows local contamination levels to more accurately reflect individual pesticide exposure. Consistent with this observation, European Robins also showed clear responses to pesticide pressure in gardens where contamination decreased with increasing distance from the nearest conventional field. Although home range sizes may vary with habitats and environmental conditions, European Robins appear to be a reliable indicator species of local contamination consistently in both farmlands and gardens, whereas other species such as Common Nightingales may reflect pesticide exposure over broader spatial scales (Becker, 2003; Siddig et al., 2016). Indeed, this distinction raises a broader consideration regarding the spatial scale at which blood pesticide contamination reflects true local exposure, particularly for migratory species. Most passerine species considered here, including migratory ones, are known to show strong fidelity to their breeding sites as adults (unpublished personal data, Feng et al., 2019; Kwon et al., 2022; Sedgwick, 2004). Although blood reflects relatively recent exposure (Espín et al., 2016), the lag time for an individual to become representative of its local environment after settling remains unknown, and likely varies with the persistence and elimination rate of each compound. For instance, in a first study at the same site, malathion, an insecticide banned in France since 2008 and across Europe since 2020, was frequently detected in Common Nightingales (Bailly et al., 2025). Such detections may arise from several non-mutually exclusive sources: (i) the continued or illicit local use of a banned compound, (ii) its residual persistence in the environment, or (iii) exposure acquired outside the study area, as this species winters in parts of Africa where malathion remains authorised. Although the latter hypothesis is plausible for a long-distance migrant, the available data do not allow these sources to be disentangled. Migratory passerines such as the Common Nightingale and the Melodious Warbler could nonetheless serve as complementary bioindicators of pesticide contamination and regulatory heterogeneity at broader geographic scales, beyond the local landscape captured by more sedentary species such as the European Robin. This is consistent with recent evidence identifying wildlife migration as one pathway for long-range transboundary pesticide transport, alongside surface and groundwater flows and atmospheric dispersion, and highlighting heterogeneous international regulation of pesticide use and residue limits as a key driver of such transboundary contamination (Tang et al., 2025). Future studies combining pesticide biomonitoring with movement tracking (e.g., GPS or radio-tracking), stable isotope analysis, repeated sampling across the annual cycle, or other tissues integrating contamination over different exposure windows (e.g., feathers) would help disentangle local exposure from contamination acquired during migration. Such tracking data would also provide direct estimates of species- and population-specific home ranges and foraging distances during the breeding season, which currently rely on literature-based approximations rather than measurements from the individuals included in our study. Nevertheless, given the limited number of species and individuals considered per species, these analyses need to be continued but offer promising avenues for future research. Therefore, it is necessary to include more species, increase sample sizes, and continue using phylogenetic approaches to better understand the importance of home range, as well as other ecological traits such as diet and migratory status, in the contamination patterns observed in passerines.

5. Conclusion

In summary, this study provides clear evidence of the widespread pesticide contamination affecting non-target wildlife. Our results

demonstrate that both organic farming and increasing distance from treated fields can significantly reduce pesticide contamination in passerine birds. This reinforces the value of certain passerine species as bioindicators, as their contamination reliably reflects environmental pollutant pressure. However, despite these mitigating factors, our findings raise concerns about persistent environmental contamination and contamination of non-target organisms, and highlight that increasing the size of regulatory buffer zones, combined with an overall reduction in pesticide use, would be necessary to effectively limit residents' exposure. More broadly, if pesticide exposure in mobile wildlife can reflect contamination acquired across administrative boundaries, this suggests that pesticide pollution should not be regarded solely as a local agricultural management issue, but also calls for coordinated regional, national, and international efforts in pesticide monitoring, regulation, and environmental risk reduction. Future research should now focus on the sublethal effects of pesticides on avian life-history traits (e.g., physiology, morphology, and behaviour) to better assess risks at the individual level. Within a 'One Health' framework, the health of domestic and wild animals, humans, plants, and the environment are closely linked and interdependent (Destoumieux-Garzón et al., 2018; Giraudoux, 2023). Such adverse outcomes in birds may therefore also serve as early warning signals of potential health risks for humans. In the near future, exploring whether the reduction in contamination provided by the mitigating components (i.e., higher proportions of organic farming in farmland environments and increased distances to the nearest treated field) also applies to human populations would also be interesting. An essential approach would be to quantify pesticide residues in environmental matrices (e.g., water, soil, air, seeds) as well as in biological matrices (e.g., insects, pets, humans through hair and urine). Such data (currently under analysis) would provide crucial insights into the relative contributions of different exposure routes and enable a more comprehensive assessment of the risks of pesticide exposure and health implications for both wildlife and humans.

CRediT authorship contribution statement

Audrey Bailly: Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Karine Monceau:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Maurice Millet:** Methodology, Formal analysis, Data curation. **Etienne Debenest:** Methodology. **Sabrina Gaba:** Writing – review & editing, Methodology, Funding acquisition, Formal analysis, Conceptualization. **Vincent Bretagnolle:** Writing – review & editing, Methodology, Conceptualization. **Jérôme Moreau:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

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Declaration of competing interest

The authors 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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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2026.110574.

Data availability

Data will be made available on request.

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