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1 **Impacts of neonicotinoids on biodiversity: a critical review**

2

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7 Special Issue “Key learnings from a collective scientific assessment on the effects of plant
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9

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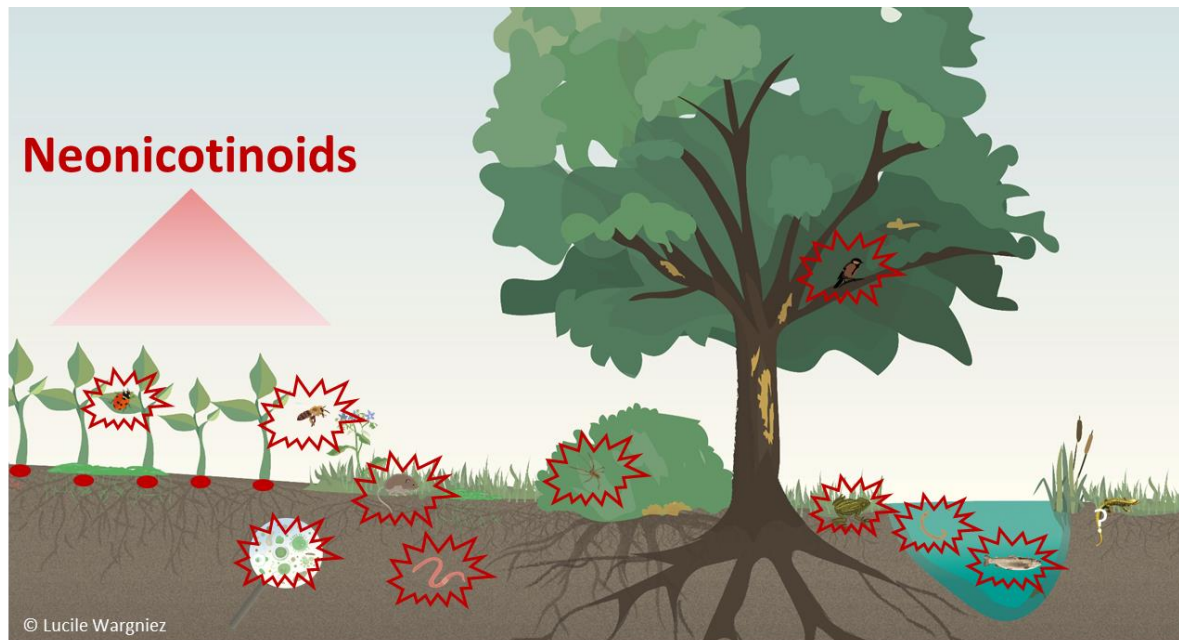
44 **Abstract**

45 Neonicotinoids are the most widely used class of insecticides in the world but they have raised numerous concerns
46 regarding their effects on biodiversity. Thus, the objective of this work was to do a critical review of the
47 contamination of the environment (soil, water, air, biota) by neonicotinoids (acetamiprid, clothianidin,
48 imidacloprid, thiacloprid, thiamethoxam) and of their impacts on terrestrial and aquatic biodiversity.
49 Neonicotinoids are very frequently detected in soils and in freshwater, and they are also found in the air. They
50 have only been recently monitored in coastal and marine environments, but some studies already reported the
51 presence of imidacloprid and thiamethoxam in transitional or semi-enclosed ecosystems (lagoons, bays and
52 estuaries). The contamination of the environment leads to the exposure and to the contamination of non-target
53 organisms, and to negative effects on biodiversity. Direct impacts of neonicotinoids are mainly reported on
54 terrestrial invertebrates (e.g., pollinators, natural enemies, earthworms) and vertebrates (e.g., birds), and on aquatic
55 invertebrates (e.g., arthropods). Impacts on aquatic vertebrate populations and communities, as well as on
56 microorganisms, are less documented. In addition to their toxicity to directly exposed organisms, neonicotinoid
57 induce indirect effects via trophic cascades as demonstrated in several species (terrestrial and aquatic

58 invertebrates). However, more data are needed to reach firmer conclusions and to get a clearer picture of such
59 indirect effects. Finally, we identified specific knowledge gaps that need to be filled to better understand the effects
60 of neonicotinoids on terrestrial, freshwater and marine organisms, as well as on ecosystem services associated with
61 these biotas.

62

63 Graphical abstract



65

66 **Keywords** Pesticides · Plant protection products · Ecotoxicity · Ecotoxicology · Agrosystems · Collective
67 scientific assessment

68

69 Introduction

70 Neonicotinoids are systemic insecticides (i.e., they diffuse throughout the treated plants to protect them from pests)
71 that act on the central nervous system of insects by targeting nicotinic acetylcholine receptors (nAChRs) in the
72 brain (Simon-Delso et al. 2015; Thompson et al. 2020). They are the world's fastest-growing and currently the
73 most widely used class of insecticides against a broad spectrum of sucking and chewing insects (plant hoppers,
74 thrips, micro-lepidopteras), and they are also involved in veterinary medicine (e.g., against fleas in pets) and in
75 biocidal products such as those used for the treatment of livestock buildings or in pest baits for domestic use
76 (Klingelhöfer et al. 2022; Thompson et al. 2020). In agriculture, neonicotinoids are mainly applied through seed

77 treatments, but they are also employed as granular application, spraying or soil treatment (Simon-Delso et al. 2015;
78 Thompson et al. 2020). The five most used active substances are acetamiprid, clothianidin, imidacloprid,
79 thiacloprid and thiamethoxam (clothianidin is also the main transformation product of thiamethoxam). Among
80 these substances, only acetamiprid is still approved in the European Union (EU Pesticides database 2023).
81 Clothianidin and thiamethoxam were withdrawn in 2019, while imidacloprid and thiacloprid were withdrawn in
82 2020 (European Commission 2023). However, for example in France, derogations have been granted in 2021 and
83 2022 for the use of coated seeds treated with imidacloprid or thiamethoxam in the context of the infestation of beet
84 crops by aphids (JORF 2021; JORF 2022). Consequently, because of their wide use all over the world, and because
85 of the high persistence of clothianidin, imidacloprid and thiamethoxam (average half-life in soils is 121 days for
86 thiamethoxam (PPDB 2023), 187 days for imidacloprid (PPDB 2023) and 545 days for clothianidin (PPDB 2023)
87 which could reach 20 years (Thompson et al. 2020)), neonicotinoids are likely to be ubiquitous in the environment,
88 and present a potential environmental health concern (Bonmatin et al. 2015; Goulson 2013; Humann-Guilleminot
89 et al. 2019a; Morrissey et al. 2015).

90 Neonicotinoids were first presented as having key attributes such as systemic nature, versatility in
91 application (especially as seed treatments), selective toxicity to arthropods, lower binding efficiencies to vertebrate
92 compared to invertebrate receptors, and assumed lower impacts on non-target aquatic and terrestrial organisms
93 (Simon-Delso et al. 2015; Thompson et al. 2020). Neonicotinoids should also theoretically not target organisms
94 lacking nAChRs and thus nervous systems, such as protists, fungi, prokaryotes and plants (Simon-Delso et al.
95 2015).

96 However, neonicotinoids appeared to have lethal and sublethal effects on non-target organisms, including
97 pollinators, insect predators and vertebrates (especially birds) (Alsafran et al. 2022; Mineau and Kern 2023;
98 Mineau and Palmer 2013; Simon-Deslo et al. 2015). Thus, for many years, the use of neonicotinoid-based products
99 in agriculture has raised concerns in several countries, particularly because of their effects on pollinators
100 (Demortain 2021; Suryanarayanan 2013), and EFSA (2018) concluded that most uses of neonicotinoid substances
101 do represent a risk to wild bees and honeybees. In addition, as more than 80% of neonicotinoid seed treatments
102 can remain in the soil (Alford and Krupke 2017; Sur and Stork 2003), soil invertebrates may be exposed to high
103 doses of neonicotinoids, with recognized lethal and sublethal effects (Gunstone et al. 2021). Neonicotinoids also
104 contaminate freshwater ecosystems worldwide and could impact aquatic invertebrates, over broad spatial scales
105 (Cavallaro et al. 2019; Hallmann et al. 2014; Morrissey et al. 2015). Moreover, they were demonstrated to exert

106 negative effects on terrestrial and aquatic vertebrates (Gibbons et al. 2015; Thompson et al. 2020; Wood and
107 Goulson 2017).

108 In this context, the objective of this work was to do a critical review of (1) the contamination of the
109 environment (soil, water, air, biota) by neonicotinoids and (2) their impacts on terrestrial and aquatic biodiversity.
110 Although the literature focused on the ecotoxicological effects of neonicotinoids is abundant, to the best of our
111 knowledge, no review has been published on the overall impacts of these substances on the whole biodiversity.

112

113 **Bibliographic corpus**

114 The review of the literature on the impacts of neonicotinoids on biodiversity was performed under the framework
115 of a French collective scientific assessment focused on the impacts of plant protection products (PPPs) on
116 biodiversity and ecosystem services (Pesce et al. 2023). Collective scientific assessment seeks to inform public
117 policy and to foster public debate by analyzing the literature, but it is neither a meta-analysis nor a systematic
118 review (Pesce et al. 2021). Though not quantitative, this review gives a detailed and complete overview of the
119 impacts of neonicotinoids on the whole biodiversity.

120 In this framework, the bibliographic corpus was adapted and constructed as follows: six queries (Q)
121 focused on neonicotinoids (Q1), ecotoxicology (Q2), biodiversity (Q3), terrestrial ecosystems (Q4), freshwater
122 ecosystems (Q5) and marine ecosystems (Q6) were defined with related keywords (Table SII). The literature
123 search was conducted on the Web of Science™, from 2000 to 2020.

124 The corpus of publications was then built by combining Q1 with Q2, Q3, Q4, Q5 or Q6. The combination
125 of Q1*Q2 yielded 7349 references; that of Q1*Q3, 457 references; Q1*Q4, 3309 references; Q1*Q5, 841
126 references; and Q1*Q6, 252 references. After removing duplicates, the total number of references was 7697.

127 The time course of the 7697 references showed a strong increase in the number of publications related to
128 the impacts of neonicotinoids from 2000 to 2020 (Fig. 1). Among the five neonicotinoids retained in this review,
129 imidacloprid was the most studied one (4218 occurrences in titles and abstracts), well above thiamethoxam (1672),
130 acetamiprid (1176), clothianidin (887) and thiacloprid (674) (Fig. 2). The bibliometric measurements also
131 demonstrated that terrestrial invertebrates were the most studied organisms and especially honeybees (Fig. 3).
132 Apart from terrestrial invertebrates, fish come at the thirty second place (Fig. 3). In the first 35 occurrences, there
133 are no other taxonomic group.

134 The categorization of references was based on titles and abstracts. The selected corpus was then divided
135 according to the expertise of the different authors who proceeded to in-depth analysis of each reference. The

136 literature search was focused on the most integrative and ecologically realistic studies as possible. The results of
137 single-species tests were not systematically reviewed, and were only used if they provided explanatory elements
138 for processes observed under realistic environmental conditions.

139 The corpus was finally manually completed by various documents, papers and books known to the authors and
140 which were not present in the 7697 references, and over time until April 2023. At the end, a total of 308
141 publications were retained and cited in this work.

142

143 **Terrestrial ecosystems**

144 **Contamination of soils, plants and air**

145 Neonicotinoids are found in all environments: soil, water (see section “Contamination of freshwater and marine
146 environments” below), plants and air.

147

148 **Contamination of soils**

149 Soil contamination by neonicotinoids has been studied under various climates, soil types, and agricultural practices
150 (Table 1). A large study conducted on 74 French cultivated soils showed that imidacloprid (limit of quantification
151 LOQ = 1 µg/kg) was present in 91% of the soil samples (excluding seven organically grown soils, with no
152 detectable traces) although only 15% of the sites had been planted with treated seeds the year of the monitoring
153 (Bonmatin et al. 2005a). In addition, imidacloprid was detected in 100% of the soils which received treated seeds
154 (corn, wheat or barley) during the sampling year, and in 97% of the soils which received the same treatment one
155 or two years before the study. Concentrations were higher in the soils which had been treated consecutively during
156 two years before the monitoring than in those that received treated seeds only one year before, indicating that
157 imidacloprid accumulates in soils over time. Silva et al. (2019) found that imidacloprid was present in 7% of the
158 examined European topsoil samples (LOQ = 10 µg/kg, one order of magnitude higher than the above study) with
159 a maximum content of 60 µg/kg, while Pelosi et al. (2021) found imidacloprid in 90 % of French sampled soils
160 (n=180, 26 % when considering concentrations >10 µg/kg, LOQ = 0.4 µg/kg) and concentrations reaching 160
161 µg/kg (Table 1). Thiamethoxam was present in 20% of the French soils at low concentrations (maximum of 2
162 µg/kg, LOQ = 0.4 µg/kg) (Pelosi et al. 2021). In Switzerland, imidacloprid (LOQ = 0.9 10⁻³ µg/kg) was quantified
163 in 94% of cultivated field soils (n=82) and in 71% of ecological focus area soils (annual, biennial and perennial
164 herbaceous plant species; n=68) (Humann-Guillemot et al. 2019a). Clothianidin (LOQ = 1.6 10⁻³ µg/kg) was

165 also frequently observed in the sampled soils (77% of cultivated fields and 46% of ecological focus areas);
166 followed by thiacloprid (LOQ = $1.6 \cdot 10^{-3}$ $\mu\text{g}/\text{kg}$; 28% and 13%), thiamethoxam (LOQ = $1.9 \cdot 10^{-3}$ $\mu\text{g}/\text{kg}$; 27% and
167 6%) and acetamiprid (LOQ = $2.0 \cdot 10^{-3}$ $\mu\text{g}/\text{kg}$; 13% and 3%) (Humann-Guillemint et al. 2019a). Similarly, Riedo
168 et al. (2021) repeatedly observed imidacloprid (59% of soils, maximum concentration of 24 $\mu\text{g}/\text{kg}$, LOQ = 0.14
169 $\mu\text{g}/\text{kg}$), clothianidin (55%, 57 $\mu\text{g}/\text{kg}$, LOQ = 0.15 $\mu\text{g}/\text{kg}$), thiamethoxam (21%, 24 $\mu\text{g}/\text{kg}$, LOQ = 0.15 $\mu\text{g}/\text{kg}$) and
170 thiacloprid (10%, 14 $\mu\text{g}/\text{kg}$, LOQ = 0.073 $\mu\text{g}/\text{kg}$) in various Swiss agricultural soils (Table 1). The highest
171 concentration of imidacloprid in Switzerland was measured by Chiaia-Hernandez et al. (2017) and was found to
172 be 138 $\mu\text{g}/\text{kg}$ (LOQ = 3 $\mu\text{g}/\text{kg}$) (Table 1). Recently, Froger et al. (2023) monitored 111 PPP residues (48 fungicides,
173 36 herbicides, 25 insecticides and/or acaricides, and two safeners) in 47 soils sampled across France under various
174 land uses (arable lands, vineyards, orchards, forests, grasslands, brownfields). The most frequently quantified
175 neonicotinoid was clothianidin (17% of the soil samples, maximum concentration of 2.7 $\mu\text{g}/\text{kg}$, LOQ = 0.5 $\mu\text{g}/\text{kg}$)
176 followed by imidacloprid (9%, 13.8 $\mu\text{g}/\text{kg}$, LOQ = 2 $\mu\text{g}/\text{kg}$), thiacloprid (6%, 0.26 $\mu\text{g}/\text{kg}$, LOQ = 0.05 $\mu\text{g}/\text{kg}$) and
177 acetamiprid (2%, 0.48 $\mu\text{g}/\text{kg}$, LOQ = 0.01 $\mu\text{g}/\text{kg}$) (Table 1). Thiamethoxam was not quantified (LOQ = 0.5 $\mu\text{g}/\text{kg}$).
178 In English arable soils, where neonicotinoids have been used as seed treatments, the concentrations of clothianidin
179 ranged from < 0.02 to 13.6 $\mu\text{g}/\text{kg}$ (LOQ = 0.02 $\mu\text{g}/\text{kg}$), that of imidacloprid from < 0.09 to 10.7 $\mu\text{g}/\text{kg}$ (LOQ = 0.09
180 $\mu\text{g}/\text{kg}$) and that of thiamethoxam from < 0.02 to 1.5 $\mu\text{g}/\text{kg}$ (LOQ = 0.02 $\mu\text{g}/\text{kg}$) (Jones et al. 2014). Overall, most
181 of the reviewed works focusing on the presence of neonicotinoids in soils is centered on imidacloprid, while the
182 other substances are much less targeted. The environmental conditions, crops, agricultural practices, analytical
183 methods and sampling time and strategies may explain the differences observed between the reviewed studies but,
184 in general, they show the ubiquitous contamination of soils by neonicotinoids (Bonmatin et al. 2015; Froger et al.
185 2023).

186

187 **Contamination of plants**

188 Neonicotinoids enter plants through the roots and/or leaves, and are transported into various organs, including
189 foliage, flowers, pollen and nectar (Bonmatin et al. 2015). They are frequently detected in cultivated plants, as
190 well as in wild plants. Bonmatin et al. (2005b) measured imidacloprid concentrations in corn whose seeds have
191 been treated with this insecticide and observed that 76% of stem and leaf samples at flowering contained more
192 than 1 $\mu\text{g}/\text{kg}$ of the substance (LOQ = 0.1 $\mu\text{g}/\text{kg}$). They also quantified from 1 to 10 $\mu\text{g}/\text{kg}$ of imidacloprid in
193 sunflower flower heads, with notable variations depending on crop stage and seed variety. In the 29 analyzed
194 samples of sunflower pollens, only two contained traces of imidacloprid. In parallel, imidacloprid was detected in

195 untreated sunflower heads grown on soil treated in previous years (from 0.1 to 2 µg/kg). In sugar beet crop treated
196 with 90 g/ha of imidacloprid as seed coating, the concentration of imidacloprid in leaves initially reached 12.4
197 mg/kg (fresh weight), then decreased but remained above 1 mg/kg 80 days after sowing, and was below the limit
198 of detection (LOD = 10 µg/kg) at harvest (Rouchaud et al. 1994). Humann-Guilleminot et al. (2019a) analyzed
199 imidacloprid, clothianidin, thiamethoxam, thiacloprid and acetamiprid in plant samples taken from 79 cultivated
200 fields (mainly from cereals and beetroots, but also from potatoes, rapeseed, maize, peas and flax) and 69 ecological
201 focus areas over Switzerland. The neonicotinoids were detected in 97% of plant samples taken in cultivated fields,
202 and in 93% of plant samples from ecological focus areas. The most frequently detected substance was imidacloprid
203 (87% in cultivated fields and 84% in ecological focus areas), followed by thiacloprid (43% and 59%), clothianidin
204 (39% and 12%), acetamiprid (34% and 45%) and thiamethoxam (19% and 7%).

205 Neonicotinoid residues were also detected in various wildflowers present in non-treated area surrounding
206 crops grown from treated seeds, with residues in foliage ranging from 0.06 to 106 µg/kg (LOQ ranged from 0.06
207 to 0.60 µg/kg) (Botias et al. 2015; Botias et al. 2016). The authors pointed that these residues may overlap with
208 lethal toxicity levels for some insect species (e.g., *Aphis glycines*). In addition, the widespread contamination of
209 wild plants in agricultural landscape likely increases the exposure duration of pollinators though it is often
210 supposed to be restricted to the crop flowering time (Botias et al. 2015).

211 Finally, in guttation droplets, potentially consumed by non-target species, works conducted in various
212 European countries showed neonicotinoid concentrations of hundreds of mg/L at the emergence of plant, but only
213 of a few µg/L one month after its emergence (Bonmatin et al. 2015; Tapparo et al. 2011).

214

215 **Contamination of air**

216 Neonicotinoids may also reach the atmosphere. The measurement of their concentrations relies on active air
217 sampling systems and by trapping compounds on a sorbent from which the compounds are extracted and analyzed.
218 Most of the time, the measured concentrations represent the sum of the compounds present in the atmosphere in
219 both particulate and gaseous forms. Désert et al. (2018) monitored PPP concentrations in ambient air samples
220 collected from February 2012 to December 2017 at one rural and six urban sites in the French Provence-Alpes-
221 Côte d'Azur region. Imidacloprid was quantified in four locations, with concentrations higher than 1 ng/m³ (LOD
222 = 0.081 ng/m³), but with a low frequency of quantification (1 to 2% depending on the site). As it was detected both
223 in the rural and urban sampling sites, the authors suggested an atmospheric transport from agricultural areas to
224 cities given the air mass retro-trajectories. In the French Phytatmo database (2023), which synthesizes the data

225 obtained by the French Approved Air Quality Monitoring Associations (AASQAs) from 2002 up to now, the
226 average imidacloprid concentration, calculated from 18 quantifications, was equal to 0.39 ng/m³, with a maximum
227 of 2.3 ng/m³ (Table 1), which was higher than the range of concentrations reported by Coscollà and Yusà (2016)
228 (from 0.012 to 0.014 ng/m³) or by Raina-Fulton (2015) (from 0.01 to 0.36 ng/m³ in the particulate phase, LOD =
229 0.0039 ng/m³) in Canada. The analysis of the Phytatmo database (2023) also showed that acetamiprid and
230 thiamethoxam were only detected once, while thiacloprid was found at an average concentration of 0.17 ng/m³ out
231 of 17 quantifications, and at a maximum concentration of 0.47 ng/m³. In Canada, for the particulate phase, Raina-
232 Fulton (2015) and Coscollà and Yusà (2016) reported acetamiprid concentrations of 0.006 ng/m³ and 0.018 ng/m³,
233 respectively, and Raina-Fulton (2015) observed clothianidin concentrations ranging from 0.01 to 0.09 ng/m³.

234

235 **Impacts on terrestrial biodiversity**

236 **Terrestrial heterotrophic microorganisms**

237 Most studies devoted to the effects of neonicotinoids on functional activities and biodiversity of terrestrial
238 heterotrophic microorganisms concerned imidacloprid. Acetamiprid, clothianidin and thiamethoxam were scarcely
239 addressed, while there was no data for thiacloprid.

240 In laboratory experiments, Cycoń and Piotrowska-Seget (2015a) evaluated the impact of imidacloprid on
241 soil microbial activities in soils spiked at the agricultural dose and at ten times this dose (1 and 10 mg/kg,
242 respectively). At the agricultural dose, imidacloprid decreased microbial respiration, total bacterial count, and
243 dehydrogenase, phosphatase and urease activities after 14 days. However, these effects were transient and the
244 measured microbial functions recovered after 56 days of exposure. At ten times the agricultural dose, imidacloprid
245 decreased the microbial parameters but no recovery was observed after 56 days suggesting irremediable impacts
246 on communities. Consistently, nitrate concentration decreased while ammonium concentration increased, in
247 agreement with the high sensitivity of nitrifying and nitrogen-fixing bacteria to imidacloprid. Under the same
248 experimental conditions, the effect of imidacloprid on the structure of ammonia-oxidizing archaea (AOA) and
249 bacteria (AOB) communities was analyzed using Denaturing Gradient Gel Electrophoresis (DGGE) (Cycoń and
250 Piotrowska-Seget, 2015b). At the agricultural dose, imidacloprid did not affect the α diversity of the bacterial
251 communities. However, at ten times the dose, imidacloprid decreased the α diversity of the AOA community in a
252 durable way, and temporarily that of the AOB community. In addition, at the highest dose, imidacloprid decreased
253 nitrification and increased ammonification. To determine the role of the microbial community diversity in the fate
254 and impact of imidacloprid and acetamiprid, Zhang et al. (2017) used soil microcosms cropped with *Brassica*

255 *chinensis* L. They showed that the diversity of the microbial community did not affect the amount of imidacloprid
256 or acetamiprid remaining in the soil but, when microbial diversity decreased, the amount of insecticide exported
257 from the soil to the plant increased. Finally, a study conducted on microbial strains isolated from soil and exposed
258 to imidacloprid or thiamethoxam in Petri dishes showed that both neonicotinoids altered the functions of *Klebsellia*
259 sp. strain 19, a phosphate-solubilizing rhizobacterium exhibiting Plant Growth Promoting Rhizobacteria (PGPR)
260 properties (Ahemad and Khan 2011). Thus, these two insecticides could compromise the PGPR activity of
261 microbial inoculant used to decrease crop dependence on chemically derived fertilizers.

262 In field conditions, soybean imidacloprid treated seeds decreased the number of *Rhizobia* by a factor of
263 three, while the number of *Rhizobia* was not affected after foliar application (Sarnaik et al. 2006). In contrast,
264 regardless of the mode of application, the insecticide had no effect on phosphate solubilizing bacteria (Sarnaik et
265 al. 2006). Li et al. (2018) studied the impact of imidacloprid or clothianidin treated seeds on the wheat rhizosphere
266 microbial communities over nine months. The analysis of 16S rRNA and ITS amplicons generated from soil-
267 extracted DNA revealed changes in the α and β diversities of bacterial and fungal communities during plant
268 development, but did not reveal any change due to seed treatment with each of the two insecticides. Furthermore,
269 under these conditions, no effect of imidacloprid or clothianidin on some biocontrol agents (*Bacillus*,
270 *Pseudomonas*, *Streptomyces*...) was observed in the wheat rhizosphere.

271 Two studies examined the impact of thiamethoxam on the taxonomic and metabolic diversity of soil
272 bacterial communities using a laboratory setting. In forest land soils spiked with different amounts of
273 thiamethoxam, an altered composition of the community was observed (Yu et al. 2020): the relative abundance of
274 *Gemmatimonadetes* and OD1 decreased when compared to the control while the relative abundance of *Chloroflexi*
275 and *Nitrospirae* increased. On the other hand, the catabolic diversity of the microbial community in soils treated
276 with the lowest dose (0.02 mg/kg) of thiamethoxam was higher than that of the control while it was lower at the
277 highest doses (0.2 mg/kg and 2 mg/kg). Analyzing soil samples from experimental plots where thiamethoxam was
278 applied in field conditions, Filimon et al. (2015) showed that the insecticide only slightly reduced the phosphatase
279 activity but reduced the number of nitrifying bacteria by about 60%.

280 In general, studies concerning the effects of neonicotinoids on terrestrial heterotrophic microorganisms
281 revealed contradictory results depending on whether they were conducted in the laboratory (often under unrealistic
282 agricultural conditions), showing impacts on the structure and on different microbial activities, or in the field (in
283 more realistic conditions), showing no or very little effect of these substances.

284

285 **Terrestrial invertebrates**

286 Neonicotinoids have negative impacts on terrestrial invertebrates (pollinators, natural enemies, earthworms...) in
287 agricultural environments despite variable responses depending on the traits and groups considered, as summarized
288 below.

289

290 ***Pollinators***

291 Neonicotinoids are likely to have greater effects on insect pollinators than other insecticides because they are
292 systemic insecticides regularly found in pollen, nectar, and other vegetative parts of plants throughout their
293 flowering period (Krupke et al. 2012; Krupke and Long 2015), leading to risks of pollinators exposure via the oral
294 route as well as through contact for a longer period of time. In addition, during their application, neonicotinoids
295 can also contaminate the surrounding environments (Krupke et al. 2012; Krupke and Long 2015). Comparative
296 toxicity studies among the different categories of neonicotinoids are scarce, but Arena and Sgolastra (2014)
297 provided some insights. They showed that nitro-substituted neonicotinoids (“N-nitroguanidines”; including
298 imidacloprid, thiamethoxam or clothianidin) were generally more toxic to pollinators than cyano-substituted
299 neonicotinoids (“N-cyanoamidines”; including acetamiprid or thiacloprid).

300 **Honeybees.** Exposure of honeybees (*Apis mellifera*) to neonicotinoids has been repeatedly demonstrated (e.g.,
301 Bonmatin et al. 2015; Hladik et al. 2016; Mitchell et al. 2017; Zhang et al. 2023). In pollens sampled in 2002-2003
302 before spring, summer, autumn and winter, in apiaries located in five French departments, imidacloprid and/or its
303 6-chloronicotinic acid transformation product were detected in 69% of the 81 samples, and quantified in 13.5%
304 and 34.6% of the samples, respectively (Chauzat et al., 2006). The frequency of detection did not vary much
305 according to the sampling period. This study was then continued until the end of 2005 (Chauzat et al. 2011):
306 imidacloprid was detected in 11.2% of the bees (average concentration of 1.2 µg/kg) and in 40.5% of the pollen
307 samples (0.9 µg/kg), and 6-chloronicotinic acid was detected in 18.7% of the bees (1.0 µg/kg) and in 33% of the
308 pollen (1.2 µg/kg). In different sites cultivated with a corn/rapeseed rotation whose seeds were treated with
309 thiamethoxam (or not), residues of thiamethoxam and clothianidin in pollens were close to the LOQ (1 µg/kg) in
310 both corn and oilseed rape (from 1 to 2 µg/kg), and the amounts in oilseed rape nectar were lower than 1 µg/kg
311 (LOQ = 0.5 µg/kg) (no corn nectar was analyzed) (Pilling et al. 2013). Wiest et al. (2011) detected imidacloprid
312 in 1% of pollen and 2% of honey but nothing in bees sampled from hives located in the French Pays de la Loire
313 region. Thiamethoxam and clothianidin were not detected in any of these samples. The multiple potential exposure

314 pathways and the size of the pollinator activity zone make it challenging to fully identify and quantify the exposure
315 of pollinators to neonicotinoids (van der Sluijs et al. 2013).

316 In parallel of the awareness raised by exposure data on the possible role of neonicotinoids in the massive
317 decline of insects, honeybees have been the subject of extensive research focused on the toxicological effects of
318 neonicotinoids. Particular concern resulted from studies focused on honeybee behavior which revealed
319 neonicotinoid-induced impairment of memory and learning abilities (Tison et al. 2019; Willemsen and Hailey
320 2001) because such impairment is likely to affect navigation parameters and the ability to return to the hive (Henry
321 et al. 2012; Henry et al. 2014). With regard to interaction with other factors or stressors, neonicotinoids were found
322 to increase the susceptibility of honeybees to pathogens (*Nosema*) (Grassl et al. 2018; Müller 2018; Pettis et al.
323 2013; Uhl and Brühl 2019). Furthermore, the effects of neonicotinoids were demonstrated to increase with
324 decrease in temperature: the ability of bees to return to the hive following exposure to thiamethoxam decreased at
325 lower temperatures (< 28°C) (Henry et al. 2014; Monchanin et al. 2019). Finally, neonicotinoids can interact with
326 other PPPs as observed for clothianidin and propiconazole (fungicide) which impact honeybee survival via
327 synergistic effects (Sgolastra et al. 2017).

328 However, the issue of the effects of neonicotinoids on honeybees has been the subject of much
329 controversy. In their large-scale monitoring study, Rolke et al. (2016) showed that honeybee colonies placed in
330 clothianidin-treated oilseed rape crops exhibited developmental and reproduction performances similar to those of
331 non-exposed colonies. Under the same crop treatment, clothianidin was not found to pose a risk to colonies in
332 terms of health, development, and overwintering success of honeybee colonies (Belsky and Joshi 2020). This result
333 was also found by Rundlöf et al. (2015) for clothianidin-rape seed treated seed in combination with non-systemic
334 pyrethroid (beta-cyfluthrin) treatments. Conversely, Samson-Robert et al. (2017) observed an increased mortality
335 of honeybee colonies located in environments dominated by clothianidin-treated grain corn. More recently, Schott
336 et al. (2021) demonstrated lethal effects of clothianidin on honeybee larvae, but found short-term resilience of
337 colonies to treatments, which may result from compensation mechanisms (increased brood size). As to adults, seed
338 treatments with clothianidin, thiamethoxam or imidacloprid resulted in increased worker bees mortality, but effects
339 on colony growth were not observed thereafter (Lin et al. 2021). Actually, the effects of neonicotinoids on colony
340 size vary across study areas (Woodcock et al. 2017). Spatial features, such as landscape characteristics and
341 especially landmarks density (landscape elements that are used as visual cues for the orientation of bees), as well
342 as the bee experience in the studied area (e.g., homing experiments carried out with foragers familiar or not with

343 the release point), influence the performance of individuals and therefore of colonies, which in turn can either limit
344 or exacerbate the neonicotinoid-induced effects (Henry et al. 2014).

345 To go further into toxicity mechanisms and their consequences for bee colony survival, LaLone et al.
346 (2017) built a network of six Adverse Outcome Pathways (AOPs) and used weight of evidence (WoE) evaluation
347 to describe plausible causal relationships between neonicotinoid mechanisms of action (activation of nicotinic
348 acetylcholine receptor as molecular initiating event and downstream molecular, cellular, or organism-level key
349 events) and colony death, as adverse outcome of regulatory concern. However, WoE assessment identified
350 uncertainty, and thereby need for further research, in some upstream-to-downstream key-event relationships (e.g.,
351 between mitochondrial dysfunction and learning/memory, or between role change in the colony and further larval
352 development).

353 **Wild bees.** Beside works on the emblematic species *Apis mellifera*, some studies have focused on wild bees. In
354 ground nesting species (*Eucera pruinosa*), soil treatment with imidacloprid was found to affect reproduction
355 (decreased number of nests and larvae) and pollen consumption whereas no effect was observed with
356 thiamethoxam used as seed treatment (*Cucurbita pepo*) (Chan and Raine 2021). However, seed treatments may
357 lead to soil contamination, even in fields adjacent to crops and in non-cropped borders, and affect native bee
358 nesting and richness (Main et al. 2020; Rundlöf et al. 2015). In the field, exposure to various neonicotinoids and/or
359 other PPPs have lethal and sublethal effects, as shown for the solitary bee *Osmia bicornis*: clothianidin or
360 thiamethoxam, used in combination with other insecticides (beta-cyfluthrin) or fungicides (fludioxonil and
361 metalaxyl-M) impaired the reproduction (Woodcock et al. 2017), as did the mixture of thiacloprid and prochloraz
362 (fungicide) (Alkassab et al. 2020), while clothianidin and propiconazole (fungicide) induced mortality (Sgolastra
363 et al. 2017). In a multistress context, the effects of neonicotinoids on wild bees can be exacerbated by food resource
364 limitation (Stuligross and Williams, 2020). Indeed, the diversification of non-crop floral resources can provide
365 complementary resources, counteracting the negative effects of neonicotinoids as shown on *O. bicornis*
366 reproduction and larval development (Klaus et al. 2021). With regard to other physiological mechanisms
367 underlying population-level responses under field conditions, the negative effects of neonicotinoids observed on
368 *Osmia cornuta* reproduction (Stuligross and Williams 2020) or at population level (fitness, density; Sandrock et
369 al. 2014) may have a male component (thiamethoxam-altered male fertility; Strobl et al. 2021a) or not
370 (clothianidin unaffected male survival, emergence and reproductive physiology; Strobl et al. 2021b). Using simple
371 generalized and linear mixed models (GLMM), Stuligross and Williams (2021) demonstrated how past and current

372 exposure to neonicotinoids profoundly impact both individual reproduction and population growth rate of orchard
373 blue bees (*Osmia lignaria*).

374 The impact of neonicotinoids on wild social bees of the *Melipona* group is very little studied. However,
375 the meta-analysis of Botina et al. (2020) highlighted lethal effects on both larvae and adults, especially marked for
376 imidacloprid.

377 In bumblebees, the effects of neonicotinoids were found to be expressed at the organism level (molecular,
378 cellular, and physiological responses; lethal and sublethal effects) as well as at the population level (mortality,
379 altered colony structure and turnover) (Camp and Lehmann 2021). Colonies of *Bombus terrestris* and *Bombus*
380 *impatiens* exposed to acetamiprid, clothianidin or imidacloprid exhibited lower growth rates and decreased
381 production of new queens (Camp et al. 2020; Rundlöf et al. 2015; Whitehorn et al. 2012). In addition, a suite of
382 effects was observed, including increased mortality of new queens, delayed nest foundation (Wu-Smart and Spivak
383 2018), acute and chronic effects on worker foraging activity (Gill and Raine 2014), reduced fecundity and brood
384 production (imidacloprid; Laycock et al. 2012), disruption of their flight activity and endurance (imidacloprid;
385 Kenna et al. 2019), and altered queen condition upon overwintering (thiamethoxam and clothianidin; Fauser et al.
386 2017). Some works also showed that seed treatments affect *Bombus* spp. densities in adjacent fields and in non-
387 cropped borders (Main et al. 2020; Rundlöf et al. 2015). With respect to interactions with other stressors, no
388 synergistic nor additive effects could be detected between neonicotinoids (mixture of thiamethoxam and
389 clothianidin) and the trypanosome parasite *Crithidia bombi* on post hibernation performances (queen survival and
390 body mass) of *B. terrestris* (Fauser et al. 2017).

391 With a multi-species dynamic Bayesian occupancy model, Woodcock et al. (2016) highlighted the high
392 impact of neonicotinoid seed treatments as use in oilseed rape on the extinction of 62 species of wild bee
393 populations. Their model was spatially and temporally explicit and related population persistence to exposure over
394 a wide time period of 18 years. This paper identifies the need of developing national scale management strategies
395 to support wild bee populations persistence over the long-term.

396 **Butterflies.** The impacts of neonicotinoids on lepidopterans are very little investigated, but the few studies
397 addressing this issue underline a critical role of the timing and mode of exposure. In the monarch butterfly (*Danaus*
398 *plexippus*), exposure of young adults to realistic doses of imidacloprid did not affect oocyte production, but
399 significantly decreased insect longevity, with likely consequences for population development, migration, and
400 overwintering (James 2019). On the contrary, under exposure to clothianidin-treated plants in the larval stage,
401 there was no significant effect on parameters characterizing monarch migration (flight orientation, movement

402 speed; Wilcox et al. 2021). Using a linear mixed effect random slope model, Gilburn et al. (2015) demonstrated
403 that the populations of 15 butterfly species commonly occurring at farmland sites in England declined due to the
404 use of neonicotinoids.

405 **Overview of the effects of neonicotinoids on pollinators.** In 2018, EFSA (2018) confirmed that the use of
406 neonicotinoids causes a risk to wild bees and honeybees. Although results appeared sometimes contradictory,
407 many studies highlighted negative effects of neonicotinoids on pollinators. The contradictions occasionally
408 observed can be explained by several methodological biases (Walters 2016): (1) laboratory experiments consider
409 exposure conditions (in particular doses and durations) to neonicotinoids that are not really representative of those
410 observed in natura in relation to agricultural practices; (2) most of the studies focus on honeybees or bumblebees,
411 whereas susceptibility to insecticides varies greatly among the different groups of pollinators (Lundin et al. 2015;
412 Rundlöf et al. 2015); (3) studies are most often focused on one type of neonicotinoid which makes generalization
413 difficult. Furthermore, there is a need to combine laboratory and field approaches, and to address the effects of
414 neonicotinoids at the sub-individual and individual levels, as well as the consequences for colonies and populations
415 (see LaLone et al. 2017). For example, Henry et al. (2015) showed that the mortality in honeybee colonies near
416 neonicotinoid (thiamethoxam and imidacloprid)-treated oilseed rape fields was higher than in colonies surrounded
417 by less treated fields. However, this effect was not observable at the colony level during and after the flowering
418 period of oilseed rape, because the impact of this loss was buffered by the colonies' demographic regulation
419 response. While very few models exist that are devoted to the effects of neonicotinoids at the bee colony/population
420 levels, this research area appears promising given the difficulty of actually detecting unintended effects of
421 neonicotinoids in the field using conventional risk assessment methods (Lundin et al. 2015). In particular, Henry
422 et al. (2017) advocated the potentialities of mechanistic models in a multiple stressor context. Since then, the
423 honeybee colony model (BEEHAVE, Becher et al. 2014) has been extended to the colony development of
424 bumblebees in a realistic landscape (Becher et al. 2018), and to translate results from standard laboratory studies
425 to relevant parameters and processes for simulating bee colony dynamics (Preuss et al. 2022). On a regulatory
426 point of view, significant efforts have been undertaken at the EU level to improve risk assessment of the effects of
427 neonicotinoid on bees with, among others, the development of the ApisRAM population model (Adriaanse et al.
428 2023; EFSA PPR Panel 2015; EFSA Scientific Committee et al. 2021).

429

430

431

432 **Natural enemies**

433 Overall, neonicotinoids have negative impacts on natural enemies such as predators (mites, ladybugs) and
434 parasitoids, especially in field crops (Douglas and Tooker 2016). By disrupting prey-predatory and host-parasitoid
435 interactions, neonicotinoid-treated seeds also alter arthropod communities as a whole (Chen et al. 2016; Disque et
436 al. 2019; Dubey et al., 2020).

437 **Ants.** In *Tetramorium caespitum*, increased mortality and disruption of locomotion without loss of hunting
438 behavior was observed after exposure to imidacloprid (Penn and Dale 2017). In other ant species (*Pogonomyrmex*
439 *occidentalis*, *Lasius niger*, *Lasius flavus*), imidacloprid was also found to alter socio-behavioral traits (e.g.,
440 foraging, nest building, competition behavior) at environmentally relevant concentrations under experimental
441 exposure (Sappington 2018; Thiel and Kohler 2016).

442 **Bugs.** Prey consumption was reduced in predatory bugs (*Pentatomidae*) feeding on herbivorous preys previously
443 exposed to imidacloprid-treated plants, even when prey density increased (lack of a type II functional response)
444 (Resende-Silva et al. 2019). Studies with *Orius insidiosus* concluded that imidacloprid was moderately to highly
445 toxic when applied as seed treatment, while foliar toxicity showed conflicting results (Naranjo 2001). In *Podisus*
446 *nigrispinus* predatory bugs, sublethal effects of thiamethoxam treatments resulted in longer larval development,
447 decreased adult body weight and delayed oviposition (Torres et al. 2003). Imidacloprid may also alter the predatory
448 behavior of spined soldier bugs (*Podisus maculiventris*), with negative consequences in terms of weight gain
449 (Resende-Silva et al. 2019). However, some of these effects were only seen at certain treatment doses (> 0.5
450 mg/plant) (Torres et al. 2003), and were sometimes transient (Pekar and Kocourek 2004).

451 **Carabids.** When fed with slugs contaminated with thiamethoxam, *Chlaenius tricolor* carabid beetles displayed
452 altered mobility twitching and mild motor difficulties, up to partial to extensive paralysis (Douglas et al. 2015).

453 **Forficulidae.** As dominant earwig species in temperate orchards, *Forficula auricularia* is the most studied
454 forficulidae species in the laboratory. Shaw and Wallis (2010) demonstrated impaired mobility and movement
455 coordination in 70 % of earwigs exposed to thiacloprid, and that more than 80 % of them died after 10 days
456 exposure. Thiacloprid was also shown to reduce larval growth and to decrease adult foraging behavior (Fountain
457 and Harris 2015). Acetamiprid significantly decreased the predation behavior of adult males by 28 % but not of
458 females nor nymphs when applied in apple orchards at the agricultural rate (Malagnoux et al. 2015).

459 **Lacewings.** Survival of the green lacewings *Chrysoperla carnea* reduced when adults feed on imidacloprid-treated
460 plants (Rogers et al. 2007). In addition, imidacloprid was found to disrupt the mobility of individuals (appearance

461 of tremors; Rogers et al. 2007). It has to be underlined that, upon multigeneration exposure, this species was able
462 to develop strong resistance to acetamiprid (Mansoor and Shad 2020).

463 **Ladybugs.** Ladybugs are impacted by neonicotinoids via prey ingestion, especially at early larval stage in
464 *Coleomegilla maculata* feeding on cereal aphids exposed to thiamethoxam (Bredeson et al. 2015). Thiamethoxam
465 reduces the mobility of ladybugs (the time to turn around when placed on their backs increases with the
466 concentration of ingested insecticide) but not the number of eggs, while a negative correlation between the increase
467 in the concentration of the insecticide and the number of developing eggs has been shown (Bredeson and Lundgren
468 2018). Wang et al. (2018a) evaluated the toxicity of thiamethoxam to *Harmonia axyridis*, a predator of the *Myzus*
469 *persicae* aphid, and its effect in term of functional response, by three exposure routes: direct contact of *H. axyridis*
470 with thiamethoxam residues; cabbage leaves infested with *M. persicae* treated systematically with thiamethoxam
471 which exposed *H. axyridis* to the insecticide indirectly (referred as systemic application, mimicking direct soil
472 drench or seed treatments); and cabbage leaves infested with *M. persicae* treated with thiamethoxam by leaf-dip
473 which exposed *H. axyridis* to thiamethoxam residues on both cabbage leaves and thiamethoxam-treated *M.*
474 *persicae* (referred as leaf dip treatment, mimicking foliar spray application). Predation was negatively affected
475 under the three conditions, but particularly when ladybugs were exposed following leaf dipping. For all exposure
476 routes, *H. axyridis* rapidly recovered predatory ability, however, sublethal effects of thiamethoxam may reduce
477 the population growth of *H. axyridis* and, therefore, impair the biological control of *M. persicae*, especially after
478 leaf or contact exposure.

479 **Parasitoid hymenoptera.** Acetamiprid was demonstrated to cause significant reductions in the abundances of
480 various groups of parasitoids (Aphelinidae, Braconidae, Encyrtidae, Eulophidae, Eupelmidae, Ichneumonidae,
481 Mymaridae, Platygasteridae, Proctotrupidae, Pteromalidae, Scelionidae, Trichogrammatidae) (Khans and
482 Alhewairini 2019), and these losses were generally accompanied by an increase in pest infestation levels (Saito et
483 al. 2008). In various parasitoid species, systemic applications of imidacloprid were often minimally detrimental,
484 whereas foliar applications could be highly toxic (Naranjo 2001).

485 **Predatory mites.** In the presence of neonicotinoids (acetamiprid, clothianidin, imidacloprid, thiacloprid or
486 thiamethoxam), disruption of mite behavior (*Panonychus ulmi*, *Tetranychus urticae*), without loss of abundance,
487 resulted in loss of biological control activity (Beers et al. 2005). Predatory mites (Phytoseiidae) are affected by
488 acetamiprid, but studies have shown that they can develop resistance (Fountain and Medd 2015) which led to a
489 growing interest in their use in sustainable agriculture (Duso et al. 2014; Fountain and Medd 2015).

490 **Spiders.** For several spider families (Araneidae, Lycosidae), contact exposure to neonicotinoids (acetamiprid,
491 imidacloprid) appeared to be the most toxic pathway (compared to consumption of treated prey) inducing lethal
492 and sublethal effects such as disruption of web construction (Pekar 2012). Furthermore, neonicotinoids
493 (acetamiprid, thiacloprid) were demonstrated to affect the richness of spider communities (Rosas-Ramos et al.
494 2020).

495

496 ***Detritivorous arthropods***

497 In a three-year field experiment, Pearsons and Tooker (2021) showed that seed treatments (corn, soybean) with
498 neonicotinoids (clothianidin, imidacloprid) reduced saprophagous arthropod (millipede, springtails, oribatid mites)
499 density and activity (litter decomposition) by more than 10%.

500

501 ***Earthworms***

502 Earthworms are likely to be exposed to neonicotinoids in soils. For example, in a French arable landscape, Pelosi
503 et al. (2021) observed residues of imidacloprid in 79% of the sampled earthworms (*Allolobophora chlorotica*,
504 n=155; maximum concentration of 777 µg/kg; 43 % of the earthworms contained imidacloprid concentrations
505 >100 µg/kg, LOQ = 0.4 µg/kg), while thiacloprid was found in 34% of the earthworms (maximum concentration
506 of 42.1 µg/kg, LOQ = 0.1 µg/kg).

507 Neonicotinoids (e.g., acetamiprid, clothianidin, imidacloprid, thiamethoxam) have negative effects on
508 several endpoints of various earthworm species (e.g., *Eisenia fetida*, *Lumbricus terrestris*, *Aporrectodea*
509 *caliginosa*), from sub-individual to community levels: tissue integrity, physiological activity, behavior, growth,
510 reproduction, and survival (Dittbrenner et al. 2010; Dittbrenner et al. 2011a; Dittbrenner et al. 2011b; Qi et al.
511 2018; Tu et al. 2011; Wang et al. 2015). They are also known to be toxic to compost worms (*E. fetida*) in laboratory
512 conditions: they affect reproduction, cellulase activity and tissues, among others (Wang et al. 2015).

513

514 ***Nematodes***

515 Compared to arthropods, nematodes tend to be less sensitive to neonicotinoids (Kudelska et al. 2017; Neury-
516 Ormanni et al. 2019; Bradford et al. 2020). In entomopathogenic species (*Steinernema glaseri*, *Steinernema*
517 *carpocapsae*, *Steinernema feltiae*, *Heterorhabditis bacteriophora*, *Heterorhabditis megidis*), a positive effect of
518 imidacloprid was observed at low dose on reproduction (Koppenhöfer et al. 2003).

519

520 **Terrestrial vertebrates**

521 ***Birds (excluding raptors)***

522 Numerous studies demonstrated that bird decline in agroecosystems is related to the use of neonicotinoids (Ertl et
523 al. 2018; Lennon et al. 2019; Li et al. 2020; Mineau and Palmer 2013; Mineau and Kern 2023).

524 In agricultural areas and other environments across Europe and North America, the analyses of
525 neonicotinoid residues in various biological components (eggs, feathers, livers, plasmas) of several avian trophic
526 groups such as nectarivores, granivores, insectivores and carnivores showed ubiquitous exposure of birds
527 (gamebirds, house sparrows, hummingbirds, songbirds...) (Bishop et al. 2020; Bro et al. 2016; Fuentes et al. 2023;
528 Humann-Guillemot et al. 2019b; Humann-Guillemot et al. 2021; Lennon et al. 2020a; Lennon et al. 2020b;
529 Poisson et al. 2021; Prouteau 2021; Roy et al. 2020). The prevalence of exposure greatly varies from one study to
530 another and among species, but, even if some studies detected neonicotinoids only in a few individuals (e.g.,
531 Graves et al. 2022), the vast majority of works underlined pervasive exposure of numerous species and pointed
532 out high frequencies of detection.

533 Granivorous birds are directly exposed to neonicotinoids following the consumption of neonicotinoid
534 treated seeds (Lopez-Antia et al. 2016; Prosser and Hart 2005; Roy et al. 2019). For example, Lennon et al. (2020b)
535 demonstrated that the detection of clothianidin in the plasma of several farmland bird species increased from 11%
536 before sowing to 51% after sowing. In French cereal dominated landscape, where neonicotinoid treated seeds were
537 widely used, the eggs or livers of grey partridge (*Perdix perdix*) and of some *Columba* species were found to be
538 contaminated by neonicotinoids (Bro et al. 2016; Millot et al. 2017). In Ontario fields (Canada), the analysis of
539 carcasses of wild turkey (*Meleagris gallopavo silvestris*), which consumes neonicotinoid-coated seeds, showed
540 detectable levels of clothianidin and/or thiamethoxam in 22.5% of individuals (detection of both substances in 5%)
541 (MacDonald et al. 2018). These studies underlined that the crop sowing periods are the most at risk (especially in
542 autumn compared to early spring, Millot et al. 2017) for bird exposure through neonicotinoid treated seeds, because
543 it also corresponds to a period of low food availability and of migration stopover for some species. Along
544 agricultural gradients in Minnesota (USA), at least one neonicotinoid among the seven compounds screened
545 (acetamiprid, clothianidin, dinotefuran, imidacloprid, nitenpyram, thiacloprid, thiamethoxam) was detected in 93
546 % and 80 % of fecal pellets of sharp-tailed grouse (*Tympanuchus phasianellus*) and greater prairie-chickens (*T.*
547 *cupido*), respectively, and in 90 % and 76 % of their livers, respectively (Roy and Chen 2023). Imidacloprid and
548 clothianidin were the most detected substances. To document the exposure of wild bird communities, Anderson et
549 al. (2023) analyzed seven neonicotinoids (acetamiprid, clothianidin, dinotefuran, imidacloprid, nitenpyram,

550 thiacloprid, thiamethoxam) in plasma samples from 55 species across 17 avian families, in four counties in Texas
551 (USA). Imidacloprid was detected in 36 % of samples (n=294), and two birds contained imidacloprid, acetamiprid
552 and thiacloprid. Clothianidin, and thiamethoxam were not detected but their LOD (0.3 µg/L, 0.05 µg/L,
553 respectively) were higher than that of imidacloprid (0.005 µg/L). Temporal variations have been evidenced, with
554 lower frequencies of detection in summer and winter than in spring and fall which correspond to the usual planting
555 days for the most common crops across the state. Some species showed higher prevalence of exposure such as the
556 American robin (*Turdus migratorius*) and the red-winged blackbird (*Agelaius phoeniceus*). Importantly, the study
557 evidenced a chronic or repeated exposure of wildlife since six birds out of seven re-sampled over time exhibited
558 at least one detection of neonicotinoid, and three exhibited multiple exposure at different time points (Anderson et
559 al. 2023). In Europe, several measurements of neonicotinoid residues in bird carcasses (livers or gizzards) revealed
560 very large numbers of accidental direct bird poisonings (passerine, *Columba* and game species) following the
561 ingestion of neonicotinoid-treated seeds, especially with imidacloprid (Berny et al. 1999; Bro et al. 2010;
562 Buchweitz et al. 2019; Millot et al. 2017; Mineau and Kern 2023; Mineau and Palmer 2013). Despite biases in the
563 detection of carcasses in the field survey (de Snoo et al. 1999; Vyas 1999), a significant number of birds have been
564 categorically identified as victims of acute and lethal poisoning induced by neonicotinoids used in seed treatments.
565 Nevertheless, these lightning mortality events would likely not be the primary cause of the significant decline of
566 some bird species (gray partridge) in agricultural environments, but they are undeniably an aggravating factor
567 (Millot et al. 2017). This is all the more since many other direct sublethal (physiological and behavioral) and
568 indirect effects of neonicotinoids have been demonstrated, for many more species than just granivores (Gibbons
569 et al. 2015; Wood and Goulson 2017). Improved seeding techniques can limit the risk of direct poisoning by
570 ensuring that treated seeds are effectively buried so that the proportion of seeds on the surface after planting is low
571 (McGee et al. 2018). However, the effectiveness of these methods depends on planting techniques and on seed
572 type and are not generalizable to all coated seed situations (McGee et al. 2018). Coatings have been suggested to
573 induce an aversion which limits ingestion to a few coated seeds, representing only a small fraction of the
574 neonicotinoid LD50 (Lethal Dose causing the death of 50% of exposed organisms) (Avery et al. 1994), but these
575 results have been shown to depend on the experimental context, including the availability of alternative food
576 resources or the state of food stress (Millot et al. 2017; Mineau and Kern 2023; Mineau and Palmer 2013).
577 Furthermore, the repellent effect results from the induction of a physiological disorder following initial ingestions
578 of treated seeds, involving that significant sublethal effects can occur well before ingestion of a lethal dose (Lopez-
579 Antia et al. 2014; Lopez-Antia et al. 2015; Mineau 2017). It has to be underlined that some passerine species,

580 especially *Fringillidae*, can de-husk seeds which lowers their direct exposure by ingestion (Prosser and Hart 2005).
581 Other contexts of neonicotinoid poisoning of passerines (American goldfinches *Spinus tristis*) have also been
582 identified in public spaces in California (Rogers et al. 2019): the mortality of birds was due to the ingestion of
583 natural elm seeds remaining on the ground which were contaminated with imidacloprid during the drench
584 application.

585 While neonicotinoids were initially thought to be less harmful to birds than insects due to their lower
586 affinity for vertebrate nicotinic receptors, mounting evidence now challenges this view and birds appear to be more
587 sensitive to neonicotinoids than other vertebrates (Mineau and Kern 2023; Mineau and Palmer 2013). The acute
588 toxicity of neonicotinoids was reported to be underestimated by a factor of ten for some wild bird species compared
589 to the one determined on model species of mallard or bobwhite quail (*Colinus virginianus*) (Mineau and Kern
590 2023; Mineau and Palmer 2013). Chronic toxicity is poorly taken into account, as well as sublethal effects which
591 are scarcely investigated.

592 Several reviews of the individual and sub-individual effects of neonicotinoids on birds have been
593 published (Gibbons et al. 2015; Moreau et al. 2022; Pisa et al. 2015; Wood and Goulson 2017). The literature
594 shows that imidacloprid induces weight loss or reduces energy reserves (fat mass) in the white-crowned sparrow
595 (*Zonotrichia leucophrys*) (Eng et al. 2017; 2019). In hummingbirds (*Selasphorus rufus*), the consumption of
596 imidacloprid in flower nectar induces underactivity and decreased energy expenditure (-25%), with no other effect
597 detected on feeding activity or immune response (Bishop et al. 2018; English et al. 2021). On the contrary, some
598 studies showed an impact of imidacloprid on the immune status of adult (Lopez-Antia et al. 2013) and juvenile
599 (Lopez-Antia et al. 2015) red-legged partridges (*Alectoris rufa*). These contrasting results could be explained by
600 interspecific variability and various exposure conditions (dose x species x biomarkers x duration) (English et al.
601 2021; Gibbons et al. 2015; Lopez-Antia et al. 2015). Behavioral alterations were also observed (Eng et al. 2019),
602 and disruption of flight and/or navigation efficiency emerged as a sensitive and relevant endpoint of imidacloprid
603 exposure and sublethal effect on the white-crowned sparrow (Eng et al. 2017). These effects have been associated
604 with loss of energy reserves. Thus, even if transient under the tested conditions, these sublethal effects can likely
605 lead to impaired migration success of white-crowned sparrows using agricultural environments as staging areas
606 (Eng et al. 2017; 2019). Furthermore, reductions in feeding and activity most often resulting in weight loss and
607 risk to survival have been demonstrated in migratory birds exposed to sublethal doses of imidacloprid (Eng et al.
608 2017; 2019). Finally, exposure to sublethal dose of acetamiprid has been associated to reduced sperm density in
609 the house sparrow (*Passer domesticus*) (Humann-Guillemint et al. 2019c).

610 In controlled experiments on red-legged partridges (*Alectoris rufa*) fed with control seeds or seeds treated
611 with imidacloprid at 20%, 100% or 200% of the recommended dose, analyses in livers showed an increase in the
612 accumulation of imidacloprid with exposure time, and mortality of 50% of the females within five days even at
613 agricultural or lower doses (Lopez-Antia et al. 2013; Lopez-Antia et al. 2015). Moreover, breeding investment was
614 lowered with reduced clutch size, eggs size and fertilization rate, and chick survival was diminished when birds
615 were exposed to imidacloprid.

616 Sabin and Mora (2022) performed an ecological risk assessment to evaluate the potential effects of
617 neonicotinoids (acetamiprid, clothianidin, imidacloprid, thiamethoxam) on populations of the northern bobwhite
618 (*C. virginianus*) in the South Texas Plains Ecoregion (USA). The assessment of the exposure of both juveniles and
619 adults showed levels which can induce adverse effects on growth, reproduction success, and long-term survival.

620 The analysis of the literature thus demonstrated that neonicotinoids are one of the factors responsible of
621 the decline in the abundance and diversity of birds. Depending on the bird species and their diet, this impact results
622 mainly either from a direct effect (e.g., ingestion of treated seeds), or from an indirect effect (e.g., reduction in
623 food resources following the decline of prey). Such indirect effects are addressed hereafter in the “Food webs”
624 section.

625

626 ***Raptors***

627 Several works showed the presence of neonicotinoids in raptors. Imidacloprid was detected in the blood of
628 Eurasian eagle owl (*Bubo bubo*) in Spain (Taliensky-Chamudis et al. 2017), imidacloprid and thiacloprid in the
629 blood of honey buzzards (*Pernis apivorus*) in Finland (Byholm et al. 2018), and acetamiprid, clothianidin,
630 thiacloprid, and thiamethoxam in the feathers of barn owls (*Tyto alba*) in Switzerland (Humann-Guilleminot et al.
631 2021). The detection frequencies were contrasted: 3% of the analyzed samples were positive in the Eurasian eagle
632 owl, whereas in the insectivorous honey buzzard, imidacloprid and thiacloprid were detected in 40 and 70% of the
633 samples, respectively. In the barn owl, more than 80% of the individuals were positive, notably for thiacloprid, the
634 frequent detection in chicks suggesting a trophic exposure. The feeding specialization of the barn owl on insects
635 would not be sufficient to explain the high detection frequency of neonicotinoids. In northern Germany, Badry et
636 al. (2021) investigated the impregnation of the livers of three raptor species (red kite *Milvus milvus*, common
637 buzzard *Buteo buteo*, Montagu’s harrier *Circus pygargus*; n=186). Among the neonicotinoids, only thiacloprid
638 was detected in two red kites. Recently, no neonicotinoid was detected in the blood of chicks of the same three

639 raptor species in Germany (Badry et al. 2022). No study examining the toxicity of neonicotinoids on raptors has
640 been identified.

641

642 ***Mammals (excluding chiropterans)***

643 One of the largest mammalian studies conducted to date resulted in the simultaneous analysis of 480 substances
644 in muscle of 42 wild boars (*Sus scrofa*), 79 roe deer (*Capreolus capreolus*) and 15 deer (*Cervus elaphus*) in Poland
645 (Kaczynski et al. 2021). The five neonicotinoids were among the most frequently detected compounds
646 (imidacloprid and thiacloprid showing mean concentrations in the top five values). They were detected in 100%
647 of the wild boar samples, while acetamiprid was detected in three deer, and thiacloprid and clothianidin were
648 detected in two deer. Acetamiprid, clothianidin and thiacloprid were detected in 13 roe deer (16.5%). The mean
649 residue concentrations ranged from 0.6 µg/kg (thiamethoxam) to 4.3 µg/kg (imidacloprid) in the liver. In France,
650 multi-residues analyses targeting 140 PPPs (67 withdrawn and 73 currently used PPPs) and transformation
651 products were performed in hair samples of small omnivorous rodents (wood mouse *Apodemus sylvaticus*) and
652 insectivorous shrews (greater white-toothed shrew *Crocidura russula*) sampled in arable landscapes (Fritsch et al.
653 2022). Again, acetamiprid, imidacloprid and thiacloprid were among the most frequently detected substances
654 (more than 80% of individuals) and/or quantified at high concentrations (up to 70.7 µg/kg) (Fritsch et al. 2022).
655 The ubiquity of exposure to neonicotinoids was demonstrated as residues were detected in all animals regardless
656 of the type of habitat (hedgerows, cereal crops, grasslands) or of the agricultural practices (conventional or organic
657 farming) (Fritsch et al. 2022). Assessing the exposure of wild raccoons (*Procyon lotor*) captured in Hokkaido
658 (Japan) to neonicotinoids (acetamiprid, imidacloprid, clothianidin, dinotefuran, thiacloprid, thiamethoxam, and
659 desmethyl-acetamiprid), Shinya et al. (2022) showed that either one of the six screened neonicotinoids or one
660 transformation product was detected in the urine of 90% of the raccoons. Neonicotinoids were also found in the
661 hair of red fox (*Vulpes vulpes*) in Italia; acetamiprid, clothianidin, and imidacloprid being detected in 100% of the
662 analyzed individuals (n=11), and thiacloprid in 91% of them (Picone et al. 2023).

663 The toxicity of neonicotinoids to mammals have been reviewed by Tomizawa (2004) and Gibbons et al.
664 (2015), showing the potential for various deleterious effects on growth, development and reproduction as well as
665 other sub-lethal effects such as genotoxic and cytotoxic effects, immunotoxicity, neuro-behavioral disorders and
666 changes in behaviors related to anxiety and fear, impairments of the thyroid and retina, and reduced movement.
667 The study of the effects of imidacloprid (112 and 225 mg/kg, daily gavage for 60 days, which is above realistic
668 environmental exposure concentrations) on rat reproduction, a mammal model organism, showed a decrease in

669 sperm vitality and number, a reduction in sex organ mass, and a decrease in the production of sex hormones FSH
670 and LH in males (Nafaji et al. 2010; Tetsatsi et al. 2019). A significant impact of imidacloprid on the rat body
671 weight was also reported but no published evidence of reproductive disorders in relation to neonicotinoid exposure
672 in wild mammals was found. However, most of the research on mammals have been performed on rats or mice,
673 and under laboratory conditions, hampering the assessment of direct toxicity to wild mammals which may exhibit
674 different sensitivity and may be exposed to other chemical or biological stressors. As for birds, Gibbons et al.
675 (2015) emphasized that neonicotinoids can also impact terrestrial mammals via indirect effects which are reviewed
676 in the “Food webs” section.

677

678 ***Chiropterans***

679 The exposure of wild bats to clothianidin, imidacloprid and thiamethoxam was demonstrated through the detection
680 of the three substances in the hair of big brown bats (*Eptesicus fuscus*) sampled in Missouri (USA) (Hooper et al.
681 2022). Imidacloprid showed the highest frequency of detection and was found in all samples (Hooper et al. 2022).
682 In Turkey, in a large screening targeting 322 PPPs and organic contaminants in adult bat carcasses of *Pipistrellus*
683 *pipistrellus* and *Myotis myotis*, 87 compounds were detected but they didn't include neonicotinoids (Kuzukiran et
684 al. 2021). Habitat preferences of these bats (urban and forest species) may limit their exposure to neonicotinoids.
685 Several studies mention a risk of exposure of chiropterans to neonicotinoids by the trophic route, based on the
686 monitoring of chiropteran activities and dosages in their prey present in the foraging sites (Stahlschmidt and Brühl
687 2012; Stahlschmidt et al. 2017).

688 In rare experimental studies, Hsiao et al. (2016) and Wu et al. (2020) reported the neurotoxic effects of
689 imidacloprid (at 20 mg/kg/day) on the echolocation ability of insectivorous bats (*Hipposideros terasensis*).
690 Memory loss in bats has been associated with apoptosis lesions in certain areas of the hippocampus (Hsiao et al.
691 2016). Another study supports these behavioral data and suggests that altered echolocation movements likely
692 affects bat movement and hunting activities (Wu-Smart and Spivak 2018). In addition, neonicotinoid use appears
693 to be associated with an increased frequency of white-nose syndrome, caused in chiropterans by a fungal infection,
694 in both the USA and Europe (Bayat et al. 2014; Oliveira et al. 2021). Upon awakening, bats experience a massive
695 inflammatory response phase with destruction of part of the immune tissue before reconstruction making them
696 particularly vulnerable to infection (Mineau and Callaghan 2018). Neonicotinoids can thus come as an aggravating
697 factor during this critical period. In their review, Mineau and Callaghan (2018) concluded that there is sufficient
698 evidence to support the assert that bats are being negatively affected by neonicotinoids, directly through functional

699 impairment, and indirectly through reduction in insect abundance (trophic cascades are detailed in the “Food webs”
700 section): the levels of neonicotinoid residues in the environment are high enough to put bats at risk of motor
701 impairment and death. Knowledge remains currently too incomplete to be able to thoroughly characterize the
702 impacts of neonicotinoids on chiropterans.

703

704 ***Reptiles***

705 Neonicotinoids (imidacloprid and thiamethoxam) have been detected in several Mongolian racerunner (*Eremias*
706 *argus*) organs and tissues (blood, brain, heart, lungs, stomach, intestine, liver, kidney, skin, fat, and gonads),
707 showing different internal distributions and post-exposure temporal variations depending on the substance
708 considered. However, the limited number of individuals which were analyzed prevents any attempt at
709 generalization (Wang et al. 2018b; Wang et al. 2019).

710 The exposure of *E. argus* to thiamethoxam and imidacloprid under controlled conditions led to variations
711 in thyroid, stress or sex hormone levels, endocrine gland damage, or changes in expression of genes involved in
712 endocrine functions (Wang et al. 2019; Wang et al. 2020). Yang et al. (2020) also reported the endocrine disrupting
713 effect of imidacloprid to *E. argus* with decreased levels of testosterone and estradiol in plasma. Further research
714 is required to better characterize the impacts of neonicotinoids on reptiles.

715

716 ***Amphibians***

717 Amphibians are one of the biological groups most affected by the collapse of biodiversity on a planetary scale, in
718 particular because of the use of PPPs (Hayes et al. 2010). However, the number of studies of the effects of
719 neonicotinoids on the terrestrial stages of amphibians is low. Comparing dermal exposure of *Hyla gratiosa* and
720 *Hyla cinerea* to imidacloprid via direct exposure of the frog present on the soil at the time of insecticide spraying,
721 and via indirect exposure following soil contact after application, van Meter et al. (2015) showed that cumulative
722 concentrations and bioconcentration factors were significantly higher for the direct exposure. In the Pampa region
723 of Argentina, imidacloprid was detected in the terrestrial *Leptodactylus latinasus* frog living in close association
724 with row crops (soybean, corn, wheat) (Brodeur et al. 2022).

725 Thompson et al. (2022) used both aquatic mesocosms, and terrestrial locomotor and behavior trials to
726 study the effects of sublethal exposure of the wood frog (*Rana sylvatica* or *Lithobates sylvaticus*) to imidacloprid.
727 The results showed a decrease in larval survival to metamorphosis under imidacloprid exposure in interaction with
728 shorter hydroperiod. However, the effect of imidacloprid depends on the frog stage: terrestrial locomotor

729 performances were improved following aquatic exposure of the larvae, while an important loss in these
730 performances was observed after terrestrial exposure to imidacloprid. In addition, high effects on population sex
731 structure and sexual development were observed: a skewed juvenile sex ratio was evidenced in imidacloprid
732 treatments with about 10% fewer males than in controls, and 15.7% of individuals exposed to imidacloprid could
733 not be assigned to either sex (ambiguous reproductive organ morphology) (Thompson et al. 2022). A great deal of
734 research remains to be done.

735

736 **Aquatic ecosystems**

737 **Contamination of freshwater and marine environments**

738 **Freshwater environment**

739 Neonicotinoids used in agricultural fields can enter surface waters (from rivers to lakes) through spray drift, dust
740 from coated seeds, runoff, subsurface flow (for example, subsurface tile drainage), input of treated leaves, and/or
741 plant decomposition in water (Alford and Krupke 2019; Stehle et al. 2018; Wang et al. 2023). The primary routes
742 of transfer are direct contamination due to spray drift or to dust abrasion of coated seeds, and re-distribution from
743 surface runoff or subsurface drainage (Schaafsma et al. 2019; Wettstein et al. 2016). Neonicotinoids are stable in
744 water, and because of their high mobility, they are mainly transported in the dissolved phase (Bonmatin et al. 2015;
745 Morrissey et al. 2015; PPDB 2023).

746 After neonicotinoid applications, the delivery ratio to surface water was estimated to be less than 2% for
747 thiamethoxam and clothianidin together, and 0.48% for imidacloprid (Frame et al. 2021; Wettstein et al. 2016).
748 The detection rates in surface water are higher after seed treatment than after spraying (Wettstein et al. 2016). In
749 North America, clothianidin was found before, during and after planting (i.e., in 98% of the samples), while the
750 detection of thiamethoxam mainly occurred in the post-plant season (54% of the samples), and that of imidacloprid
751 during the planting season (48% of the samples) (Evelsizer and Skopec 2018). Clothianidin is both a PPP and a
752 transformation product of thiamethoxam which could explain its higher frequency of detection (Wang et al. 2023).

753 Neonicotinoids have been quantified in various types of surface waters including wetlands, ditches, ponds
754 and rivers (Table 1). Acetamiprid, imidacloprid, and thiamethoxam are the most frequently detected substances
755 (Pietrzak et al. 2019). Overall, maximum concentrations of neonicotinoids in surface waters were found to be 9.14
756 µg/L for imidacloprid, 6.90 µg/L for thiamethoxam, 4.00 µg/L for acetamiprid, 3.50 µg/L for clothianidin, and
757 1.37 µg/L for thiacloprid (Alford and Krupke 2019; de Araújo et al. 2022; Criquet et al. 2017; Evelsizer and Skopec

2018; Kuechle et al. 2019; Nélieu et al. 2021; Pietrzak et al. 2019; Schaafma et al. 2019; Wang et al. 2023) (Table 1). Most of these reported maximum concentrations exceed the ecological thresholds for neonicotinoid water concentrations (0.2 µg/L for short-term acute exposure and 0.035 µg/L for long-term chronic exposure) which were defined to avoid lasting effects on aquatic invertebrate communities (Morrissey et al. 2015). A recent review provided a meta-analysis of neonicotinoid concentrations in water, based on more than 40 papers published in ten countries (Wang et al. 2023). It reported mean concentrations of 0.222 µg/L (n=1056) for clothianidin, 0.120 µg/L (n=879) for imidacloprid, 0.059 µg/L (n=863) for thiamethoxam, 0.023 µg/L (n=428) for acetamiprid, and 0.011 µg/L (n=295) for thiacloprid.

Some mitigation measures could consist in improving the application material to prevent dust during planting of treated seeds, and to improve water interception of surface and subsurface flow thanks to buffer zones such as wetlands. For example, in constructed wetlands, removal of neonicotinoids due to direct accumulation in macrophytes and to enhanced biodegradation was estimated to range from 10 to 100% in 28 days (Liu et al. 2021; Main et al. 2017).

771

772 **Marine environment**

Neonicotinoids have only been recently monitored in coastal and marine environments. Consequently, data are just available for imidacloprid and thiamethoxam, which are generally searched for using passive integrative POCIS samplers or directly in water. In mainland France, these substances were not found in the Channel/North Sea coast (Menet-Nedelec et al. 2018). On the contrary, on the other two maritime facades (Bay of Biscay and Mediterranean), imidacloprid and thiamethoxam were quantified quite frequently (with maximum frequencies of detection of 20%) in the coastal waters of the Arcachon Basin (maximum of 0.14 µg/L and 0.0039 µg/L for imidacloprid and thiamethoxam, respectively, in spot samples) (Auby et al. 2011; Tapie and Budzinski 2018) (Table 1), in transitional waters of the Gironde estuary (maximum imidacloprid concentration of 0.0053 µg/L with integrative sampling) (Levesque et al. 2018), in Marennes-Oléron bay (maximum of 0.0238 µg/L and 0.0004 µg/L for imidacloprid and thiamethoxam, respectively, with integrative sampling) (Pepin et al. 2017), and in Mediterranean lagoons (maximum of 0.028 µg/L and 0.0025 µg/L, for imidacloprid and thiamethoxam, respectively with integrative sampling) (Munaron et al. 2020; Munaron et al. 2023). Imidacloprid has also been detected in the Charente estuary and in the Loire estuary since 2006 (GIP Loire Bretagne 2013). According to ecotoxicological data collected in the OBSLAG (Observatory of the Mediterranean Lagoons) study, only imidacloprid would cause a chronic risk for the biota of lagoon ecosystems (exceeding its chronic marine predicted

788 no effect concentration PNEC in several lagoons since the beginning of the monitoring in 2017) (Munaron et al.
789 2022). This risk can be extended to the Arcachon basin and Marennes-Oléron bay given the reported data. No
790 neonicotinoid was found in French marine sediments and no reference from the French overseas territories
791 mentions their research in the water of the marine environment.

792 Only scarce information is available evidencing the contamination of marine waters worldwide. In the
793 Queensland region of Australia, streams flowing into the marine waters of the Great Barrier Reef were found to
794 be contaminated with imidacloprid at levels ranging from 0.0005 to 1.3 µg/L (Warne et al. 2022). The
795 contamination concerned observation sites located in downstream sectors near the mouths of large rivers (Warne
796 et al. 2022). This pattern appeared similar in the Bohai Sea (China), where Naumann et al. (2022) observed the
797 seasonal variation in neonicotinoid concentrations in rivers and marine water. In their study, the detection
798 frequency of acetamiprid was 100% in both river (n=72) and marine (n=81) waters in summer and fall. Despite
799 dilution in the coastal environment, the risk quotient associated with the contamination levels were reported as
800 high risk for marine organisms regarding imidacloprid, thiamethoxam and acetamiprid (Naumann et al. 2022).
801 Due to their slow degradation rates in the environment and binding properties to particulate organic matter (PPDB
802 2023), neonicotinoids are likely to accumulate in sediments: Chen et al. (2022) reported contamination of marine
803 sediments in East China Sea, due to the Yangtze River inputs, several tenths of kilometer from the river mouth.
804 The mean concentration of total neonicotinoids was 11.9 µg/kg (dry weight). The authors concluded that marine
805 sediments were a major sink for neonicotinoids, highly used in continental China as PPPs (Chen et al. 2022).

806

807 **Impacts on aquatic biodiversity**

808 **Aquatic microorganisms**

809 Few studies have been published on the effects of neonicotinoids on aquatic microorganisms. They suggest that
810 imidacloprid does not affect the activity and respiration of aquatic microbial decomposers (Kreutzweiser et al.
811 2007; Kreutzweiser et al. 2008). With the exception of the study of Neury-Ormanni et al. (2020a), who observed
812 that an exposure of the freshwater diatoms *Planothidium lanceolatum* and *Gomphonema gracile* to 5 µg/L
813 imidacloprid resulted in indirect effects via competition and predation, effects of neonicotinoids on different
814 microalgae (e.g., *Desmodesmus subspicatus*; Malev et al., 2012) and cyanobacteria (e.g., *Synechocystis* sp.; Li et
815 al., 2010) were only observed at very high concentrations (i.e., several mg/L), irrelevant to environmental
816 contamination levels. Using a quantitative structure activity-toxicity modeling approach, Gökçe and Saçan (2019)
817 also predicted an absence of effects of acetamiprid on microalgae exposed to up to 100 mg/L. Neonicotinoids are

818 therefore unlikely to be toxic to aquatic microbes, including primary producers, except under extreme events of
819 contamination.

820

821 **Aquatic invertebrates**

822 Works focused on the effects of neonicotinoids on aquatic invertebrates are increasingly investigated (as compared
823 to other insecticide classes, such as carbamates and organophosphates) due to the relative recentness of their use
824 (first homologations date back from the 1990s), and to the risk specifically posed to aquatic invertebrates because
825 of the levels of water contamination reported (see above). Morrissey et al. (2015) highlighted strong evidence that
826 water-borne neonicotinoid exposure is frequent, long-term and at concentrations which commonly exceed several
827 existing water quality guidelines. In addition, several monitoring studies of watercourses in either agricultural or
828 urban landscapes demonstrated a significant contamination of freshwater amphipods (*Gammarus pulex*) by
829 neonicotinoids (e.g., Shahid et al. 2018a; Švara et al. 2021).

830 Despite awareness of these contamination levels, works devoted to the effects of neonicotinoids on
831 aquatic invertebrate biodiversity are still limited. A first review published in 2015 noted the weak level of
832 knowledge available on the effect of neonicotinoids on the invertebrate fauna of freshwater and marine
833 environments (Pisa et al. 2015). Since then, various field case studies have provided data and
834 documented/predicted effects of neonicotinoids on aquatic invertebrate communities. For example, in Canadian
835 wetlands near treated rapeseed crops, a correlation was established between neonicotinoids (acetamiprid,
836 clothianidin, imidacloprid, thiamethoxam) transfer during rainfall events and changes in emergent insect (Diptera)
837 diversity (Cavallaro et al. 2019). Through an experimental rice mesocosm study, imidacloprid was found to
838 significantly reduce populations of various insects (dragonfly, bug, beetle) (Kobashi et al. 2017). A drastic decline
839 in zooplankton biomass in Japanese brackish lakes also coincided with the introduction of neonicotinoids
840 (clothianidin, imidacloprid, thiamethoxam) in rice agriculture since the 1990s, followed by collapse of predator
841 fish populations (Yamamuro et al. 2019). In the Netherlands, where imidacloprid residues in water are particularly
842 high, correlations between these residues and decline in arthropod taxa such as mayflies, odonates, diptera, and
843 some crustaceans were revealed on a national scale (van Dijk et al. 2013). This was also observed in a study
844 adopting a PAF (Potentially Affected Fraction) approach, but with much lower proportions of species potentially
845 affected by neonicotinoids taking into account the co-occurrence of other PPPs in the studied environments (Vijver
846 and van den Brink 2014).

847 Comparing recorded or predicted concentrations of neonicotinoids in the aquatic environment to
848 ecotoxicity thresholds has raised some concerns for the potential effects of these insecticides in freshwater
849 environments. The review by Sánchez-Bayo et al. (2016) reported widespread effects of neonicotinoids on aquatic
850 species in the USA, and the major risk for aquatic invertebrates was reaffirmed in 2017 (Wood and Goulson 2017).
851 More recently, a study based on an agricultural region located in an ecologically important wetland (Nebraska's
852 Rainwater Basin, USA), showed negative correlations between neonicotinoid concentrations and
853 macroinvertebrate biomass (which represents potential resources for various migratory birds) despite
854 concentrations below the acute toxicity risk thresholds proposed by the USEPA (Schepker et al. 2020).

855 Long-term ecological impact of neonicotinoids is a particularly salient issue for aquatic invertebrates. The
856 chronic risk mainly results from the ability of neonicotinoids to reach aquatic environments (high solubility in
857 water) and to persist there when they are adsorbed on particles (Armbrust and Peeler 2002). However, this risk is
858 poorly assessed because most often based on toxicity tests on *Daphnia*, an organism more tolerant than insects and
859 other arthropods to neonicotinoids (Beketov and Liess 2008; Wood and Goulson 2017). Neonicotinoids can have
860 chronic effects on abundance and community structure of freshwater arthropods and other macroinvertebrates at
861 doses in the $\mu\text{g/L}$ range and below (Beketov and Liess 2008; Kattwinkel et al. 2016). After cessation of treatments,
862 the onset of delayed effects was also demonstrated in situ (limnocorrals) for much lower concentrations of
863 imidacloprid and clothianidin ($< 0.05 \mu\text{g/L}$) resulting in a significant advancement of the emergence date of
864 chironomids and zygopteran odonates (Cavallaro et al. 2018; Williams and Sweetman 2019). From a functional
865 point of view, the desynchronization of phenology of these organisms could have important consequences on
866 ecosystems, especially in terms of biomass input to the terrestrial environment (trophic resource for terrestrial
867 predators such as birds). Lethal and sublethal effects of thiacloprid have been demonstrated in various aquatic
868 invertebrates, several days after exposure, for moderate acute toxicity concentrations (Beketov and Liess 2008).
869 Neury-Ormanni et al. (2020b) documented altered feeding behavior in chironomids exposed to environmental
870 doses of imidacloprid. The insecticide induced changes in motility, feeding selectivity, and browsing ability. The
871 reduced abundance and altered emergent aquatic insect assemblages in wetlands exposed to neonicotinoids could
872 explain the reduction in densities of insectivorous birds in such environments (Cavallaro et al. 2019).

873 Investigating the idea of long-term impact of neonicotinoids beyond the lifespan of exposed individuals,
874 recent works with the model amphipod crustacean, *G. pulex*, suggested the development of tolerance towards
875 clothianidin within populations from watercourses in agricultural landscapes (Becker and Liess 2017; Becker et
876 al. 2020; Shahid et al. 2018b). According to the authors, in these populations, the evolution of resistance by natural

877 selection could be facilitated by factors acting at the population and/or community levels: distance from non-
878 tolerant populations, which would favor selection locally by limiting gene flow and the influx of non-adapted
879 genes into populations (Hoffmann and Willi 2008), and low community diversity which would intensify intra-
880 specific competition in gammarids. Nevertheless, the shift in sensitivity of this non-target species to the
881 neonicotinoid appeared very moderate (less than three-fold change in LC50 for example) in comparison to the
882 genetic resistance reported for other neurotoxic insecticides (pyrethroids and organophosphates) in the amphipod
883 *Hyalella azteca* (Gamble et al. 2023; Weston et al. 2013). In addition, an inverse pattern with increased sensitivities
884 of long-term exposed *G. pulex* populations towards imidacloprid was found in non-agricultural context presenting
885 complex mixture of organic contaminants (Švara et al. 2021). Overall, these results demonstrate the unsuspected
886 importance of evolutionary adaptive processes underway in natural populations unintentionally exposed to
887 neonicotinoids, and the urgency to develop assessment tools specifically focused on long-term effects (Oziolor et
888 al. 2016). Such processes should be anticipated, at least in insects and probably in other arthropods, from the
889 current knowledge on the selective evolution of resistance to neonicotinoids in pests, based either on target-site
890 mutation or on metabolic resistance (Bass et al. 2015).

891 Although environmentally less realistic than field approaches, experimental studies performed in
892 mesocosms and in the laboratory (e.g., common garden), offer the statistical power required to test patterns
893 observed in natura (Barmantlo et al. 2021), as well as interactions with other environmental factors susceptible to
894 alleviate or aggravate the effects of neonicotinoids, such as PPP mixtures (Sanchez-Bayo and Goka 2012; Rico et
895 al. 2018; Sol Dourdin et al. 2023), temperature/climate (Mohr et al. 2012; Sumon et al. 2018; Rico et al. 2018),
896 nutrients/fertilizers (Barmantlo et al. 2019; Chara-Serna et al. 2019), vegetation disturbance (Cavallaro et al. 2019),
897 and indirect effects between species representative of different functional groups in the community (e.g., such as
898 predator-prey relationships; Miles et al. 2017). In this regard, Alexander et al. (2013) used artificial streams to
899 examine the impact of mixing three insecticides expected to act additively, i.e., imidacloprid (which acts on the
900 acetylcholine receptor) and two organophosphates which act on the acetylcholine esterase enzyme, chlorpyrifos
901 and dimethoate, and under oligotrophic vs mesotrophic (nitrate input), along a Toxic Unit (TU) gradient
902 established for concentrations consistent with environmental data. The study showed a significant interaction
903 between insecticides and nutrients on macroinvertebrate communities, with notably, under mesotrophic condition
904 and low insecticides pressure, an increase in the total abundance and species richness of ephemeropteran,
905 plecopteran and trichopteran insects. At higher insecticides pressure, the overall density of these groups and the
906 entire community was the most reduced in mesotrophic streams. In contrast, for other species groups such as

907 chironomids, detritus feeders, and the odonate predator *Gomphus* spp., no significant interaction between
908 insecticides and nitrate was detected. In oligotrophic environments, increasing PPP doses decreased predation
909 intensity, which in turn affected abundance patterns while, in mesotrophic environments, a bottom-up effect of
910 nutrients on the periphyton explained the variation in macroinvertebrates abundance and richness. Such cause-
911 and-effect relationships were also analyzed with Structural Equation Modeling (SEM) approaches which describe
912 effect pathways among different variables of interest (Miller et al. 2020; Schmidt et al. 2022). At low doses, the
913 toxicity of PPPs appeared hidden by nutrients because of increased compensatory consumption, expression of
914 adaptive plasticity at the intraspecific level, or differential responsiveness across taxa, processes which are not
915 captured by traditional community study methods (taxonomic determination and records of relative abundances).
916 Interactions between nutrients and PPP can thus result in a redirection of energy within food webs towards non-
917 productive pathways (Davis et al. 2010) or in a shift in communities towards more tolerant groups (Vinebrooke et
918 al. 2004). This type of interactions was also studied in terms of convergence/divergence of invertebrate community
919 structure in open artificial ditches (naturally assembled communities), by combining NPK elements with
920 thiacloprid (Barmantlo et al. 2019). Following thiacloprid treatments designed to maintain concentrations for one
921 month (two spikes separated by two weeks), no effect of treatments, other than an increase in total abundance after
922 four months due to nutrient input, was found in terms of taxon richness, overall abundance, or within-treatment
923 community divergence/convergence through time (β dispersion). However, significant changes were observed in
924 community composition under the effect of thiacloprid, nutrients and combination thereof. This effect persisted
925 several months after the disappearance of thiacloprid from the medium. The main compositional changes were a
926 reduction in the abundance of insects and large predators, and an increase in multivoltine species. Some results,
927 such as the particularly strong increase in *Helophorus* beetles under nutrients and thiacloprid, may reflect a PPP-
928 induced rippling effect on the community amplified by nutrient supply. This study shows that thiacloprid, in
929 addition to its short-term toxicity, induces indirect longer-term ecological effects.

930 Overall, the corpus analyzed pointed to a marked impact of neonicotinoids on aquatic arthropods at low
931 doses, as demonstrated once again in a recent study which reports the decline in emerging aquatic insects during
932 a three-month semi-field experiment considering environmentally realistic contamination scenarios of thiacloprid
933 (Barmantlo et al. 2021). However, more studies remain to be performed to determine the relationship between the
934 impacts of neonicotinoids and fitness of organisms, in relation to the ecological functions to which they contribute,
935 as well as on the relationship between the impacts of neonicotinoids on the nervous system and the behavior of
936 aquatic invertebrates.

937 **Aquatic vertebrates**

938 ***Amphibian larvae and tadpoles***

939 The sensitivity of amphibian species to neonicotinoids through water contamination has been rarely studied. Green
940 frog (*Rana clamitans*) tadpoles were found to be relatively insensitive to imidacloprid with mortality observed
941 after 96h of exposure to high concentrations only (150 mg/L) (Puglis and Boone 2011). This lack of sensitivity is
942 likely due to differences in the vertebrate nicotinic acetylcholine receptor relative to their invertebrate homologs
943 (Li et al. 2016). On the contrary, spotted marsh frog tadpoles (*Limnodynastes tasmaniensis*) suffered high mortality
944 rates (up to 17%) when they were exposed to imidacloprid concentrations as low as 0.50 µg/L (Sievers et al. 2018).
945 This exposure level reduced swimming speed and distance, and escape responses which then made the tadpoles
946 more susceptible to predation, while increasing erratic swimming (Sievers et al. 2018). The toxicity of imidacloprid
947 has also been demonstrated in the tadpoles of *Leptodactylus luctator* and *Physalaemus cuvieri* (Samojeden et al.
948 2022). The consequences of exposure to environmental concentrations (3-300 µg/L) led to a decrease in size, to
949 morphological malformations (for the two species), and to changes in tadpole swimming activity (only for *L.*
950 *luctator*).

951 In the current literature, there is limited evidence of the effects of neonicotinoids on amphibians under
952 chronic exposure to aquatic environmental concentrations. However, neurotoxic responses can be observed.
953 Campbell et al. (2022; 2023) demonstrated the ability of imidacloprid to cross the blood-brain barrier and to
954 concentrate over 300-fold in the brain of juvenile northern leopard frogs (*Rana pipiens*) with some consequences
955 on foraging behavior (e.g., a decrease in reaction times to a food stimulus by 1.5 to 3.2 times for organisms exposed
956 to concentrations up to 10 µg/L). At concentrations ranging from 0.1 to 10 µg/L and over a 21 day exposure period,
957 bioaccumulation of imidacloprid in frog brains is accompanied by a decreased reactivity in individuals subjected
958 to feeding stimuli. Beyond the active substance, the transformation product imidacloprid-olefin was detected in
959 the brains of amphibians at much lower concentrations, which does not mean that this compound cannot be
960 responsible for any toxic action. Surprisingly, exposure of leopard frogs to imidacloprid led to increased growth
961 primarily affecting body length (Campbell et al. 2022). Recent research has further demonstrated that wood frogs
962 (*R. sylvatica* or *L. sylvaticus*) exposed to imidacloprid (10 or 100 µg/L) at the tadpole stage were less likely to
963 escape simulated predator attacks in the laboratory, suggesting that exposure to this insecticide may negatively
964 impact tadpole perception and cognitive function (Lee-Jenkins and Robinson 2018; Sweeney et al. 2021).
965 However, at a lower concentration of 0.1 µg/L, imidacloprid did not induce any modulation of acetylcholinesterase
966 activity in bullfrog (*Lithobates catesbeiana*) tadpoles after three weeks of exposure (Rios et al. 2017). For other

967 less studied neonicotinoids as clothianidin, frog tadpoles are among the least sensitive species in case of
968 laboratory exposure at sublethal concentrations (Miles et al. 2017). The tadpoles are tolerant to clothianidin,
969 confirming the low toxicity of neonicotinoids in vertebrates (Miles et al. 2017). As stated in the section focused
970 on the impacts of neonicotinoids on amphibians during their terrestrial life, numerous research remain to be done
971 to characterize their impacts on amphibians in aquatic media.

972

973 ***Fish***

974 In general, neonicotinoids exhibit low acute toxicity to fish. The 96h LC50 of clothianidin ranges from 93.6 mg/L
975 for sheepshead minnow (*Cyprinodon variegatus*) to 117 mg/L for bluegill sunfish (*Lepomis macrochirus*)
976 (Anderson et al. 2015). A similar trend is observed for imidacloprid, with 96h LC50 ranging from 211 mg/L for
977 rainbow trout (*Oncorhynchus mykiss*) to 280 mg/L for common carp (*Cyprinus carpio*) (Anderson et al. 2015).
978 Two formulations of thiamethoxam have 96h LC50 above 100 mg/L (Anderson et al. 2015). These results indicate
979 that fish are insensitive to neonicotinoids, probably because of the properties of the vertebrate nicotinic
980 acetylcholine receptor (Li et al. 2016).

981 Nevertheless, the available data indicate that exposure of aquatic vertebrates to sublethal concentrations
982 of neonicotinoids results in pro-oxidative responses from which genotoxic perturbations arise. A short 48h
983 exposure of the freshwater cichlid fish (*Australoheros facetus*) to imidacloprid concentrations of 100 and 1000
984 µg/L affected the integrity of fish erythrocyte DNA (COMET assay and micro-nuclei test) (Iturburu et al. 2018).
985 Under short-term exposure to a much lower concentration of thiamethoxam (3.75 µg/L), the siluriform catfish
986 (*Rhamdia quelen*) showed activity inhibition of two liver enzymes, adenylate kinase and pyruvate kinase, as early
987 as 24h of exposure (Baldissera et al. 2018). These inhibitions were associated with a decrease in ATP levels in the
988 liver. The energetic deregulation appeared to persist after the fish were no longer contaminated (Baldissera et al.
989 2018). Beyond these non-specific effects, neonicotinoids can act on the nervous function of non-target organisms,
990 given their mode of action (binding to nicotinic acetylcholine receptors at neuromuscular junctions leading to
991 insect paralysis) (Kimura-Kuroda et al. 2012). Imidacloprid was found to be neurotoxic to adult rainbow trout (*O.*
992 *mykiss*) exposed for 21 days to high concentrations (10 and 20 mg/L) (Topal et al. 2017). This neurotoxicity
993 resulted in inhibition of acetylcholinesterase activity, oxidative stress, and a concomitant increase in DNA damage
994 in the fish brains (Topal et al. 2017).

995 Neurotoxicity of neonicotinoids may also impact the behavior of fish. A laboratory test developed to
996 investigate two key responses of fish anti-predator behaviors revealed that zebrafish (*Danio rerio*) larvae exposed

997 for 24 hours to acetamiprid exhibited increased fear reflex and faster habituation compared to unexposed larvae
998 (Faria et al. 2020). The concentrations tested in this study were considered to be realistic (0.04 and 0.40 $\mu\text{g/L}$) in
999 relation to measured concentrations of acetamiprid in surface water (0.008 to 44 $\mu\text{g/L}$) (Faria et al. 2020). The
1000 modulations of fish larvae anti-predator behavior observed in the laboratory raise questions about the
1001 environmental reality of such effects and about their hypothetical consequences in terms of survival capacity in
1002 the environment. Könemann et al. (2021) observed that zebrafish larvae were able to avoid imidacloprid
1003 contamination, but did not react to other neonicotinoids such as thiacloprid. In addition, the experimental ablation
1004 of olfaction abolished aversive responses of individuals, indicating that fish may sense insecticides. In this species,
1005 the assessment of neural activity in 289 different brain regions revealed a particular modulation of hypothalamic
1006 areas involved in the fish stress response, indicating that the observed behavioral patterns are close to those
1007 observed for other stress responses (Könemann et al. 2021). Juvenile medaka (*Oryzias latipes*), exposed to
1008 imidacloprid under rice cultivation field conditions, were consecutively infected by a *Trichodina* parasite
1009 (Sánchez-Bayo and Goka, 2005). Such pathology was linked to the chemical stress induced by imidacloprid. If
1010 toxicity of imidacloprid to vertebrates was extensively studied, the toxicity related to imidacloprid transformation
1011 products (5-hydroxy-imidacloprid, imidacloprid-urea and 6-chloronicotinic acid) was not taken into account until
1012 now, despite their presence in various tissues as observed, for example, in muscle, gonads, brain and gills in
1013 Goldfish (*Carassius auratus*) (Xu et al. 2023).

1014 A few studies deal with the combined effects of neonicotinoids with other PPPs but sometimes with
1015 experimental approaches that are more or less relevant in the context of ecological risk assessment. Thus, adult
1016 zebrafish exposed by immersion during 24 hours to high concentrations of imidacloprid (13.75 mg/L) associated
1017 with the organophosphate insecticide dichlorvos (7.5 mg/L) and the herbicide atrazine (1.5 mg/L) showed high
1018 levels of lipid peroxidation, particularly in the liver, compared to fish exposed to the same active substances tested
1019 in isolation (Shukla et al. 2017). Although this type of study is useful to test the hypothesis of expected synergistic
1020 effects, it does not allow estimation of the actual environmental risk, particularly in view of the contamination of
1021 surface waters reported by the authors (in the Ebro River in Spain: minimum concentration of imidacloprid of
1022 0.0016 $\mu\text{g/L}$ and maximum concentration of 0.015 $\mu\text{g/L}$) (Shukla et al. 2017). It is therefore important to consider
1023 such data with caution when assessing the ecotoxicity of neonicotinoids. Similarly, mixture of the order of mg/L
1024 imidacloprid and organophosphate insecticide triazophos used to assess embryotoxicity to zebrafish early larvae
1025 (blastula stage: 2h post-fertilization) exposed during 96h revealed a strong synergistic effect in terms of acute
1026 toxicity (Wu et al. 2018). Although relevant in terms of mixture toxicity assessment, such high concentrations still

1027 lack environmental relevance. It is worth noting that, though concentrations were still high, synergistic effects
1028 were also demonstrated on zebrafish larvae (72h post-hatching) for various combinations of imidacloprid with
1029 atrazine, butachlor, chlorpyrifos or lambda-cyhalothrin (mixtures containing from two to five substances) (Wang
1030 et al. 2017).

1031 No study has been devoted to the effects of neonicotinoid mixtures on aquatic vertebrates (Anderson et
1032 al., 2015). In addition, there is a lack of ecosystem-scale studies (mesocosm approaches and/or field studies) to
1033 investigate the effects of these insecticides. Work is also needed on sub-lethal or chronic effects to reflect
1034 environmental concentration levels. Finally, most of the studies focus on imidacloprid, with very little attention
1035 paid to the effects of other neonicotinoids.

1036

1037 ***Aquatic birds***

1038 Aquatic birds include waterbirds, which live in freshwater environments, and seabirds, which feed on the resources
1039 of seas and oceans.

1040 The exposure of seabirds to neonicotinoids (acetamiprid, clothianidin, imidacloprid, thiacloprid,
1041 thiamethoxam) was characterized by analyzing residues in feathers sampled from the piscivorous Sandwich tern
1042 (*Thalasseus sandvicensis*) and the mixotrophic Mediterranean gull (*Ichthyaetus melanocephalus*) in fledglings
1043 from the Lagoon of Venice (Distefano et al. 2022). Neonicotinoids were detected in both species, and imidacloprid
1044 and clothianidin were the most often quantified ones (100% in Mediterranean gulls and 58% in Sandwich terns,
1045 and 100% in Mediterranean gulls and 61% in Sandwich terns, respectively). The detection of thiacloprid was lower
1046 (<20% of samples in both species) (Distefano et al. 2022). On the contrary, no residue of neonicotinoids was found
1047 in the liver or blood of white-tailed sea eagles (*Haliaeetus albicilla*) and ospreys (*Pandion haliaetus*) (Badry et al.
1048 2021; Badry et al. 2022).

1049 For waterbirds, data are even more scarce. In some rice-growing regions, aquaponic practices involve
1050 ducks for the control of weed and pest in rice fields (Mburia, 2016). In this very particular context, ducks may be
1051 contaminated with neonicotinoid residues (Khidkhan et al., 2022).

1052 To date, no result on the direct effects of neonicotinoids on seabirds and waterbirds were available in the
1053 literature. Thus, even if the toxicity of neonicotinoids to aquatic vertebrates is presumed to be limited, there are
1054 still many areas of knowledge that need to be clarified and completed such as toxicity of transformation products,
1055 and levels of impregnation of agricultural wetland-living organisms by native substances and their transformation
1056 products (Frank and Tooker, 2020).

1057 **Food webs**

1058 Neonicotinoids can affect terrestrial and aquatic biodiversity by spreading through food webs, by the propagation
1059 of adverse biological effects in food webs and disturbance of trophic interactions (e.g., reduced predation rate,
1060 increased mortality of predators), and/or by reducing food resources (Alsafran et al. 2022). However, the number
1061 of results which have been published in the literature remains limited.

1062

1063 **Terrestrial ecosystems**

1064 Focusing on insects, Tooker and Pearsons (2021) reviewed the mechanisms underlying the effects of insecticides
1065 on food webs. They highlighted how neonicotinoids influence trophic interactions and food webs, and contribute
1066 to insect declines. Neonicotinoids spread across trophic levels, primary and secondary consumers being exposed
1067 through several routes (including dietary and trophic routes), and they may also bioaccumulate in some organisms
1068 (Tooker and Pearsons 2021). Neonicotinoids distort food webs by significantly decreasing insect abundance and
1069 diversity of both preys and consumers, as evidenced in various ecosystems (e.g., croplands, woodlands,
1070 watercourses). Depopulated and less diversified insect communities lead to food scarcity for their predators,
1071 thereby adversely impacting their local population dynamics. Importantly, food web disruption can occur even
1072 when neonicotinoids do not bioaccumulate or biomagnify in food webs, depending on the sensitivity of the taxa
1073 constituting the lower trophic levels (i.e., toxic effects on prey inducing adverse effects on higher levels via trophic
1074 cascades) and/or the sensitivity of higher trophic levels (i.e., relatively low concentrations but high enough to
1075 induce toxic effects on sensitive predators) (Tooker and Pearsons 2021).

1076 In terrestrial invertebrates, thiamethoxam has been reported to have no effect on the predation rates of
1077 two predators, *Orius insidiosus* insidious flower bug and *Hippodamia convergens* ladybug, after consuming aphids
1078 reared on thiamethoxam-treated plants (Esquivel et al. 2020). On the contrary, insidious flower bug survival, unlike
1079 that of ladybugs, was reduced following aphid consumption. However, the reduction in bug survival was only
1080 observed in the first few weeks after thiamethoxam application, and no reduction was noted one month after
1081 treatment or beyond. In an urban context (Central Park, New York City, USA) where trees were treated with
1082 imidacloprid against an alien beetle (*Anoplophora glabripennis*), unexpected outbreaks of a formerly innocuous
1083 herbivore, *Tetranychus schoenei* (Tetranychidae), followed insecticide applications to elms (Szczepaniec et al.
1084 2011). Changes in the structure of arthropod communities sampled in elm canopies after imidacloprid treatments
1085 were evidenced, mainly related to an increase in the abundance of *T. schoenei*. Laboratory tests showed that

1086 exposure to imidacloprid through consumption of imidacloprid-treated elm foliage enhanced the fecundity of *T.*
1087 *schoenei* by 40%: adult *T. schoenei* fed leaves from treated elms laid more eggs than when fed with leaves from
1088 untreated elms (Szczepaniec et al. 2011). However, no effect of imidacloprid on *T. schoenei* fecundity was detected
1089 when mites were directly sprayed with the insecticide. The longevity of mites was also not affected by exposure
1090 to imidacloprid via food. Two model predators of spider mites, the Coccinellidae *Stethorus punctillum* (adult) and
1091 the Chrysopidae *Chrysoperla rufilabris* (larva), showed significant decrease in feeding rates when offered mites
1092 from imidacloprid-treated elms as preys. Moreover, the predators exhibited signs of intoxication (partial or
1093 complete lack of response to touch, tremors, regurgitation, excessive grooming, and inability to right themselves
1094 when placed on their back) and deleterious effects when exposed to imidacloprid by consuming prey from leaves
1095 of treated trees such as impaired mobility and reduced longevity (about one-two days when mites fed from treated
1096 trees versus 9-13 days when *T. schoenei* fed from untreated trees) (Szczepaniec et al. 2011). By stimulating
1097 reproduction of mites while poisoning insect predators of spider mites which may reduce top-down regulation,
1098 imidacloprid tree treatments finally led a non-target innocuous herbivore to reach a pest status (Szczepaniec et al.
1099 2011). This study underlined how neonicotinoids may disrupt ecosystem functioning and impair ecological balance
1100 that ultimately can favor pest outbreaks. Studying the effect of thiamethoxam on the spider mite (*Tetranychus*
1101 *urticae*, considered as a pest in various agricultural systems) and its predator *Phytoseiulus persimilis*, Pozzebon et
1102 al. (2011) showed that the neonicotinoid was toxic to both *T. urticae* and *P. persimilis*, but that the impact of
1103 thiamethoxam varied according to the routes of exposure. The authors demonstrated that topical exposure led to
1104 sublethal effects in predators and preys while residual and contaminated food exposures led to both lethal and
1105 sublethal effects. In addition, toxicity increased when several exposure routes were involved. By limiting exposure
1106 to thiamethoxam to ingestion of contaminated food only, the impact of the insecticide was more favorable to *P.*
1107 *persimilis* than to its prey (Pozzebon et al. 2011).

1108 The propagation of sublethal effects of neonicotinoids via trophic interactions was evidenced in a three-
1109 level food chain gathering wild strawberry (*Fragaria vesca*), wood cricket (*Nemobius sylvestris*) and nursery web
1110 spider (*Pisaura mirabili*): strawberries were treated with imidacloprid at different doses and crickets were allowed
1111 to feed on them (Uhl et al. 2015). In this tritrophic system, feeding, mass gain, thorax growth and mobility of wood
1112 crickets was reduced, and herbivory and predation diminished at sublethal imidacloprid doses in the non-target
1113 organisms (Uhl et al. 2015). The effects of thiamethoxam, applied as a soybean seed treatment, on interactions
1114 between soybeans, non-target herbivorous mollusks (pests), and predatory insects was studied in the laboratory
1115 and in the field (Douglas et al. 2015). In the laboratory, the slug *Deroceras reticulatum* was not affected by

1116 thiamethoxam, but predatory ground beetles (*Chlaenius tricolor*) which ate these slugs were affected or died in
1117 over 60% of cases. In the field, thiamethoxam seed treatments decreased the activity and density of predatory
1118 arthropods, thereby releasing slug predation and reducing soybean densities by 19% and yield by 5%. The analyses
1119 of thiamethoxam residues revealed a transfer in food webs: they showed that insecticide concentrations decreased
1120 throughout the food chain, but that levels in slugs collected in the field were still high enough to adversely affect
1121 predatory insects. According to Douglas et al. (2015), this work on the trophic transfer of thiamethoxam challenges
1122 the idea that seed treatments with neonicotinoids specifically target herbivore pests, and underscores the need to
1123 consider predatory arthropods and soil organism communities in neonicotinoid risk assessment and management.

1124 If neonicotinoids can affect vertebrates through direct effects, as reviewed above, they can also affect
1125 wildlife through a reduction in food resources (Gibbons et al. 2015). Further, the trophic transfer of neonicotinoids
1126 has been recently evidenced, especially in birds. The presence of 54 residues of PPPs or transformation products
1127 was investigated in the food bolus (insects) provided by the parents of the tree swallow (*Tachycineta bicolor*) to
1128 their chicks, in 40 Canadian farms (Poisson et al. 2021). This multi-residue analysis included seven neonicotinoids
1129 (acetamiprid, clothianidin, dinotefuran, imidacloprid, nitenpyram, thiacloprid, thiamethoxam). The results attested
1130 to the ubiquitous trophic exposure, with nearly half of the food boluses showing contamination by at least one
1131 substance, clothianidin being among the most frequently detected PPPs (9%). Mixtures of 2 to 16 PPPs, among
1132 which five (clothianidin, dinotefuran, imidacloprid, thiacloprid, thiamethoxam) of the seven neonicotinoids, were
1133 also detected in 21% of the food boluses (and 45% of the contaminated boluses). A study conducted in Switzerland
1134 reported that at least one neonicotinoid was detected in 100% of food boluses collected from Alpine swift
1135 (*Tachymarptis melba*) provisioning their nestlings, 75% of the food boluses exhibiting measurable concentrations
1136 (Humann-Guillemot et al. 2021). Both acetamiprid and thiacloprid were found, and thiacloprid showed the
1137 highest occurrence (up to 66.7%) and the highest concentrations (up to 0.6 µg/kg). Surveys on birds in the USA
1138 and Europe revealed exposure/accumulation of neonicotinoids in all trophic groups such as nectarivores and
1139 granivores, insectivores and predators including top-predators (raptors), and piscivores, strongly suggesting the
1140 occurrence of trophic transfer in food webs (Badry et al. 2021; Bishop et al. 2020; Bro et al. 2016; Byholm et al.
1141 2018; Distefano et al. 2022; Humann-Guillemot et al. 2021; Talianky-Chamudis et al. 2017). In 60 sites over a
1142 wide cereal plain in France, the bioaccumulation of several neonicotinoids has been evidenced in both
1143 granivorous/omnivorous rodents, and insectivorous shrews as well as in earthworms and carabid beetles, which
1144 were their potential preys (Pelosi et al. 2021; Fritsch et al. 2022). Finally, residues in tissues have also been detected

1145 in terrestrial invertebrates and vertebrates, including wildlife species other than granivores (which can be exposed
1146 directly via ingestion of treated seeds) as detailed in previous sections (e.g., chiropterans).

1147 Some studies highlighted the potential for neonicotinoids to negatively impact terrestrial insectivorous
1148 vertebrate abundance and diversity through indirect effects related to the reduction in quantity and quality of food
1149 resources. Such indirect effects have rarely been studied on vertebrates but Gibbons et al. (2015) showed that
1150 systemic insecticides can induce effects on wildlife via trophic cascades: the reduction in food supply related to
1151 the use of imidacloprid led to impairments in fish species.

1152 Long before major publications based on large-scale correlative analyses between PPP use and
1153 population, Tennekes and Zillweger (2010) argued that neonicotinoid contamination of surface waters in Europe
1154 was one of the factors responsible for the continental-scale decline in insect biomass, which in turn led to many of
1155 the widespread declines in birds (golden oriole *Oriolus oriolus*, northern wheatear *Oenanthe oenanthe*, starling
1156 *Sturnus vulgaris*...). This was studied by Hallmann et al. (2014) who observed that insectivorous bird populations
1157 in the Netherlands declined in areas with surface water concentrations of imidacloprid higher than 0.02 µg/L. Spatial
1158 differences in land-use changes related to agricultural intensification (urban area, natural area, cropped area,
1159 fertilizers) have been considered but they did not alter the significance of the observed effects. In the USA, Li et
1160 al. (2020) found that the increase in neonicotinoid use was related to reductions of 4% and 3% in grassland and
1161 insectivorous bird biodiversity, respectively, over 2008-2014. Such a trend was also found for non-grassland and
1162 non-insectivorous birds, with an average annual rate of reduction of 2%. Recently, Kraus et al. (2021) conducted
1163 surveys in wetlands of cropland and grassland landscapes which allowed to characterize cross-ecosystem fluxes
1164 of PPPs mediated by aquatic insect emergence, and discussed their implications for terrestrial insectivores. Aquatic
1165 insects were estimated to transfer fluxes ranging from 2 to 180 µg of total insecticides per wetland per day to the
1166 terrestrial ecosystem. Seven PPPs were detected in newly emerged insects, among which clothianidin and
1167 imidacloprid, and biomass of emerging aquatic insects was reduced up to 73% in cropland wetlands. The authors
1168 suggested that the availability of emerging adult aquatic insect prey for insectivores was reduced by insecticides,
1169 and that accumulated insecticide could be responsible for insectivore exposure to insect-borne PPPs. Along the
1170 observed gradient in PPP levels among the different wetlands, a decrease of 43% in insect emergence but an
1171 increase of 50% in insect-mediated PPP flux with increasing insecticide concentrations were reported (from 3 to
1172 577 ng of insecticide per gram of insect) (Kraus et al. 2021). In addition, the presence of these neonicotinoids also
1173 led to a reduction in insect resources for consumer invertebrates (Kraus et al. 2021). Although bioaccumulation in
1174 organisms and transfer in food webs have been demonstrated together with sublethal and lethal effects propagated

1175 along food chains, the major process involved in shaping the impact of neonicotinoids in food webs is considered
1176 as being food web simplification (Tooker and Pearsons 2021). Such indirect effect of neonicotinoids affects both
1177 prey and predator populations through trophic cascade mechanisms and feedbacks. The initial decrease in
1178 resources when lower trophic levels are directly impacted by the use of the insecticides affect the dynamics of
1179 consumer populations at higher trophic levels through food scarcity (bottom-up control). When consumers are
1180 adversely impacted either directly (toxicity) or indirectly (lack of food supply), a subsequent decrease in predation
1181 occurs, affecting the dynamics of prey populations (top-down control). Compensatory mechanisms for consumers
1182 to overcome the decrease of one or a few food resources, such as switching to other food items, hardly occur when
1183 the predator of concern are specialist species, and seemed currently hampered in the case of neonicotinoids because
1184 of their widespread use (huge spatial extent worldwide, perennial and frequent use), the ubiquity of their
1185 environmental contamination, their broad toxicity to non-target fauna, and time-cumulative toxicity (Tooker and
1186 Pearsons 2021).

1187

1188 **Aquatic ecosystems**

1189 Adverse effects of neonicotinoids can propagate through aquatic food webs via contaminated primary producers
1190 (Lima-Fernandes et al. 2019). Lima-Fernandes et al. (2019) used imidacloprid-contaminated and uncontaminated
1191 black alder tree (*Alnus glutinosa*) leaves to feed the stonefly shredder *Protonemura* sp., which were later given as
1192 prey to *Isoperla* sp. They showed that survival, body length and biomass of the shredders as well as leaf
1193 decomposition were 20% to 50% greater in the uncontaminated treatment in comparison to imidacloprid exposure.
1194 The biomass and length of predators were 11% and 4.3% higher, respectively, when fed with uncontaminated prey
1195 than when fed with imidacloprid exposed prey (Lima-Fernandes et al. 2019). Bioaccumulation of imidacloprid has
1196 been evidenced in both *Desmognathus* salamanders (*D. monticola* and *D. fuscus*) and benthic macroinvertebrates
1197 sampled from water streams adjacent to treated hemlock stands in the USA (Crayton et al. 2020), which represents
1198 a potential source of exposure for consumers at higher trophic levels. If exposure via the trophic route was likely
1199 for salamanders, several non-exclusive routes of exposure might be involved in the subsequent bioaccumulation,
1200 including dermal and dietary uptake (Crayton et al. 2020).

1201 Hayasaka et al. (2012) showed that successive applications of imidacloprid and the phenylpyrazole
1202 insecticide fipronil (also a systemic insecticide) in experimental rice fields resulted in reduced growth of medaka
1203 fish, *Oryzias latipes*, adults and fry, most likely through reduced medaka prey abundance. Indeed, the
1204 concentrations (approximately 1 to 50 µg/L) were too low to have a direct effect on fish. As indicated above, the

1205 decline of emerging insects from aquatic ecosystems towards riparian and surrounded terrestrial landscapes
1206 strongly decrease the prey availability for numerous consumers, and overall minor energy transfer across
1207 ecosystems (Kraus et al. 2021).

1208 In a Japanese lacustrine ecosystem, Yamamuro et al. (2019) demonstrated the existing relationship
1209 between decline in fishery yields and neonicotinoids. The use of neonicotinoids on watersheds since 1993
1210 coincided with an 83% decrease in average zooplankton biomass in spring, causing the smelt (*H. nipponensis*)
1211 harvest to collapse from 240 to 22 tons. Young smelts consume zooplankton crustaceans, and their decreased
1212 abundance was linked to the reduction of zooplankton biomass caused by the introduction of neonicotinoids. This
1213 study demonstrates the indirect effects of neonicotinoids along an aquatic food web through cascading effects.

1214 Waterbirds living and feeding in lakes and ponds (ducks, waders, cormorants...) may depend on aquatic
1215 invertebrates as their food source. Consequently, the depletion of this food source must necessarily affect them
1216 (Sánchez-Bayo et al. 2016). Duckling abundance is thus related to aquatic macroinvertebrate abundance, which is
1217 consistent with other studies, and collectively suggests that neonicotinoids contamination could influence duckling
1218 abundance indirectly by impacting aquatic macroinvertebrate communities (Tyler 2022). The available data
1219 indicate that the effects of neonicotinoids on aquatic bird life are indirect, as for other bird families, and are
1220 associated with the direct toxic impacts of these contaminants on invertebrates (Sánchez-Bayo et al. 2016).

1221

1222 **Conclusion**

1223 Neonicotinoids, in particular imidacloprid, and to a lesser extent thiamethoxam and clothianidin, are very
1224 frequently detected in soils and freshwaters, even several years after their use. In addition, the presence of
1225 acetamiprid, imidacloprid, thiacloprid and thiamethoxam was observed in the air. Neonicotinoids have only been
1226 recently monitored in coastal and marine environments (since 2010s), but many studies report the presence of
1227 imidacloprid and thiamethoxam in different transitional ecosystems such as Mediterranean lagoons.

1228 This contamination of the environment leads to the exposure of non-target organisms and impacts
1229 biodiversity. The ecotoxicological effects of neonicotinoids depend on the studied organisms, but this review
1230 showed that these substances have particularly high direct and indirect impacts on terrestrial invertebrates and
1231 vertebrates, and on aquatic invertebrates. The impacts on aquatic vertebrates are less documented.

1232 The effects of neonicotinoids on terrestrial heterotrophic microorganisms vary according to the
1233 conditions: in field studies, these substances have little or no effect, while in the laboratory, impacts on the structure
1234 and on different microbial activities were observed (however, the tested concentrations are sometimes unrealistic).

1235 Laboratory studies are not always environmentally relevant, but they are complementary to field approaches as
1236 they can help to understand the effects at lower levels of biological organization (sub-individual, individual) that
1237 have consequences on higher levels (populations, community) observed in the field. Although contradictory results
1238 have been noted in the literature, neonicotinoids have negative effects (mortality, mobility disturbance) at the
1239 individual level on pollinators (honeybees in particular). In addition, exposure to neonicotinoids increases the
1240 susceptibility of honeybees to diseases and pests. Despite the importance of wild pollinators and their crucial role
1241 in pollination, the number of studies focused on the impacts of neonicotinoids on this highly diverse group of
1242 organisms is very limited. Furthermore, neonicotinoids have been shown to have effects on other terrestrial
1243 invertebrates such as natural enemies, earthworms or nematodes. Neonicotinoids are also largely involved in the
1244 decline of birds. Consumption of treated seeds is mainly responsible for neonicotinoid direct poisoning, but birds
1245 could be exposed to these insecticides especially by trophic route after consumption of contaminated insects.
1246 Neonicotinoids have negative effects on bats, amphibians, and on reptiles (though available data are still scarce
1247 for this group). For aquatic invertebrates and vertebrates, the data on the effects of neonicotinoids remain limited.
1248 The available results indicate correlations between neonicotinoid concentrations and declines in arthropod taxa.
1249 Neonicotinoids seem to be not very toxic to aquatic vertebrates such as fish, but recent studies provide worrying
1250 results for amphibians. However, the number of studies remains low and few studies focused on marine organisms.
1251 In addition to their toxicity to directly exposed organisms, neonicotinoid-induced indirect effects via trophic
1252 cascades have been demonstrated to affect some species (terrestrial and aquatic invertebrates) but data are still too
1253 few to get a clear picture.

1254 This critical review highlighted numerous knowledge gaps. First, there was a lack of data regarding the
1255 effects of neonicotinoids on primary producers (although the mode of action of neonicotinoids is unlikely to result
1256 in effects; Anderson et al. 2015), aquatic heterotrophic microorganisms, wild pollinators, raptors, mammals,
1257 reptiles, amphibians, aquatic vertebrates, and on organisms in the marine environment in general. In addition: (1)
1258 the majority of studies focused on only one neonicotinoid making generalization difficult; (2) while imidacloprid
1259 is the most commonly studied neonicotinoid, data are limited for the other substances; (3) most laboratory studies
1260 do not reflect realistic and representative uses under in field application conditions; (4) very few studies consider
1261 transformation products and mixtures with other PPPs; (5) the number of studies considering the impact of
1262 neonicotinoids on high levels of biological organization (i.e., beyond individual and population) is low; (6) the
1263 effects of neonicotinoids on maintenance of pest regulation and soil functions are hardly reported; (7) there is a
1264 lack of time series to survey mid- or long-term effects as well as post-exposure effects; (8) there is a lack of data

1265 regarding the effects of neonicotinoids on ecosystem functioning and services, yet the few existing studies suggest
1266 that they might significantly alter important provision and regulation ecosystem services (Pesce et al. 2023). More
1267 research remains to be done to better characterize the impacts of neonicotinoids to protect biodiversity.

1268

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1295 **References**

- 1296 Adriaanse P, Arce A, Focks A, Ingels B, Daniela J, Del Aguila M, Ercolano V, Ferilli F (2023) Guidance on the
1297 risk assessment of plant protection products on bees (*Apis mellifera*, *Bombus* spp. and solitary bees). *EFSA*
1298 *J* 21:7989. <https://doi.org/10.2903/j.efsa.2023.7989>
- 1299 Ahemad M, Khan MS (2011) Effects of insecticides on plant-growth-promoting activities of phosphate
1300 solubilizing rhizobacterium *Klebsiella* sp strain PS19. *Pestic Biochem Physiol* 100(1):51-56.
1301 <http://dx.doi.org/10.1016/j.pestbp.2011.02.004>
- 1302 Alexander AC, Luis AT, Culp JM, Baird DJ, Cessna AJ (2013) Can nutrients mask community responses to
1303 insecticide mixtures? *Ecotoxicology* 22(7):1085-1100. <http://dx.doi.org/10.1007/s10646-013-1096-3>
- 1304 Alford AM, Krupke CH (2017) Translocation of the neonicotinoid seed treatment clothianidin in maize. *Plos One*
1305 12:e0173836. <http://dx.doi.org/10.1371/journal.pone.0173836>
- 1306 Alford AM, Krupke CH (2019) Movement of the neonicotinoid seed treatment clothianidin into groundwater,
1307 aquatic plants, and insect herbivores. *Environ Sci Technol* 53(24):14368-14376.
1308 <http://dx.doi.org/10.1021/acs.est.9b05025>.
- 1309 Alkassab AT, Kunz N, Bischoff G, Pistorius J (2020) Comparing response of buff-tailed bumblebees and red
1310 mason bees to application of a thiacloprid-prochloraz mixture under semi-field conditions. *Ecotoxicology*
1311 29(7):846-855. <http://dx.doi.org/10.1007/s10646-020-02223-2>
- 1312 Alsafran M, Rizwan M, Usman K, Saleem MH, Al Jabri H (2022) Neonicotinoid insecticides in the environment:
1313 A critical review of their distribution, transport, fate, and toxic effects. *J Environ Chem Engineer* 10:108485.
1314 <https://doi.org/10.1016/j.jece.2022.108485>
- 1315 Anderson JC, Dubetz C, Palace VP (2015) Neonicotinoids in the Canadian aquatic environment: A literature
1316 review on current use products with a focus on fate, exposure, and biological effects. *Sci Total Environ*
1317 505:409-422. <http://dx.doi.org/10.1016/j.scitotenv.2014.09.090>
- 1318 Anderson MJ, Valdiviezo A, Conway MH, Farrell C, Andringa RK, Janik A, Chiu WA, Rusyn I, Hamer SA (2023)
1319 Imidacloprid exposure is detectable in over one third of wild bird samples from diverse Texas ecoregions.
1320 *Sci Total Environ* 876:162723. <https://doi.org/10.1016/j.scitotenv.2023.162723>
- 1321 Arena M, Sgolastra F (2014) A meta-analysis comparing the sensitivity of bees to pesticides. *Ecotoxicology*
1322 23(3):324-334. <http://dx.doi.org/10.1007/s10646-014-1190-1>
- 1323 Armbrust KL, Peeler HB (2002) Effects of formulation on the run-off of imidacloprid from turf. *Pest Manage Sci*
1324 58(7):702-706. <http://dx.doi.org/10.1002/ps.518>

- 1325 Auby I, Bost CA, Budzinski H (2011) Régression des herbiers de zostères dans le Bassin d'Arcachon : état des
1326 lieux et recherche des causes. Rapport Ifremer RST/LER/AR/11.007, 195.
- 1327 Avery ML, Decker DG, Fischer DL (1994) Cage and flight pen evaluation of avian repellency and hazard
1328 associated with imidacloprid-treated rice seed. *Crop Protec* 13(7):535-540. [http://dx.doi.org/10.1016/0261-](http://dx.doi.org/10.1016/0261-2194(94)90107-4)
1329 [2194\(94\)90107-4](http://dx.doi.org/10.1016/0261-2194(94)90107-4)
- 1330 Badry A, Schenke D, Treu G, Krone O (2021) Linking landscape composition and biological factors with exposure
1331 levels of rodenticides and agrochemicals in avian apex predators from Germany. *Environ Res* 193:110602.
1332 <http://dx.doi.org/10.1016/j.envres.2020.110602>
- 1333 Badry A, Schenke D, Brücher H, Chakarov N, Grünkorn T, Illner H, Krüger O, Marczak T, Müskens G, Nachtigall
1334 W, Zollinger R, Treu G, Krone O (2022) Spatial variation of rodenticides and emerging contaminants in
1335 blood of raptor nestlings from Germany. *Environ Sci Pollut Res* 29:60908-60921.
1336 <https://doi.org/10.1007/s11356-022-20089-1>
- 1337 Baldissera MD, Souza CF, Golombieski JI, Seben D, Sippert LR, Salbego J, Zanella R, Baldisserotto B (2018)
1338 Thiamethoxam induced hepatic energy changes in silver catfish via impairment of the phosphoryl transfer
1339 network pathway: Toxicological effects on energetics homeostasis. *Environ Tox Pharm* 60:1-4.
1340 <https://doi.org/10.1016/j.etap.2018.04.002>
- 1341 Barmantlo SH, Schrama M, van Bodegom PM, de Snoo GR, Musters CJM, Vijver MG (2019) Neonicotinoids and
1342 fertilizers jointly structure naturally assembled freshwater macroinvertebrate communities. *Sci Total Environ*
1343 691:36-44. <http://dx.doi.org/10.1016/j.scitotenv.2019.07.110>
- 1344 Barmantlo SH, Schrama M, de Snoo GR, van Bodegom PM, van Nieuwenhuijzen A, Vijver MG (2021)
1345 Experimental evidence for neonicotinoid driven decline in aquatic emerging insects. *Proc Natl Acad Sci USA*
1346 118:e2105692118. <http://dx.doi.org/10.1073/pnas.2105692118>
- 1347 Bass C, Denholm I, Williamson MS, Nauen R (2015) The global status of insect resistance to neonicotinoid
1348 insecticides. *Pestic Biochem Physiol* 121:78-87. <https://doi.org/10.1016/j.pestbp.2015.04.004>
- 1349 Bayat S, Geiser F, Kristiansen P, Wilson SC (2014) Organic contaminants in bats: Trends and new issues. *Environ*
1350 *Int* 63:40-52. <http://dx.doi.org/10.1016/j.envint.2013.10.009>
- 1351 Becher MA, Grimm V, Thorbek P, Horn J, Kennedy PJ, Osborne JL (2014) BEEHAVE: a systems model of
1352 honeybee colony dynamics and foraging to explore multifactorial causes of colony failure. *J Appl Ecol*
1353 51:470-482. <https://doi.org/https://doi.org/10.1111/1365-2664.12222>

1354 Becher MA, Twiston-Davies G, Penny TD, Goulson D, Rotheray EL, Osborne JL (2018) Bumble-BEEHAVE: A
1355 systems model for exploring multifactorial causes of bumblebee decline at individual, colony, population and
1356 community level. *J Appl Ecol* 55:2790-2801. <https://doi.org/https://doi.org/10.1111/1365-2664.13165>

1357 Becker JM, Liess M (2017) Species diversity hinders adaptation to toxicants. *Environ Sci Technol* 51(17):10195-
1358 10202. <http://dx.doi.org/10.1021/acs.est.7b02440>

1359 Becker JM, Russo R, Shahid N, Liess M (2020) Drivers of pesticide resistance in freshwater amphipods. *Sci Total*
1360 *Environ* 735:139624. <http://dx.doi.org/10.1016/j.scitotenv.2020.139264>

1361 Beers EH, Brunner JF, Dunley JE, Doerr M, Granger K (2005) Role of neonicotinyl insecticides in Washington
1362 apple integrated pest management. Part II. Nontarget effects on integrated mite control. *J Insect Sci* 5:16.
1363 <http://dx.doi.org/10.1093/jis/5.1.16>

1364 Beketov MA, Liess M (2008) Acute and delayed effects of the neonicotinoid insecticide thiacloprid on seven
1365 freshwater arthropods. *Environ Tox Chem* 27(2):461-470. <http://dx.doi.org/10.1897/07-322r.1>

1366 Belsky J, Joshi NK (2020) Effects of fungicide and herbicide chemical exposure on Apis and non-Apis bees in
1367 agricultural landscape. *Front Environ Sci* 8:81. <http://dx.doi.org/10.3389/fenvs.2020.00081>

1368 Berny PJ, Buronfosse R, Videmann B, Buronfosse T (1999) Evaluation of the toxicity of imidacloprid in wild
1369 birds. A new high performance thin layer chromatography (HPTLC) method for the analysis of liver and crop
1370 samples in suspected poisoning cases. *J Liq Chrom Rel Technol* 22(10):1547-1559.
1371 <http://dx.doi.org/10.1081/jlc-100101750>

1372 Bishop CA, Moran AJ, Toshack MC, Elle E, Maisonneuve F, Elliott JE (2018) Hummingbirds and bumble bees
1373 exposed to neonicotinoid and organophosphate insecticides in the Fraser Valley, British Columbia, Canada.
1374 *Environ Tox Chem* 37(8):2143-2152. <http://dx.doi.org/10.1002/etc.4174>

1375 Bishop CA, Woundneh MB, Maisonneuve F, Common J, Elliott JE, Moran AJ (2020) Determination of
1376 neonicotinoids and butenolide residues in avian and insect pollinators and their ambient environment in
1377 Western Canada (2017, 2018). *Sci Total Environ* 737:139386.
1378 <http://dx.doi.org/10.1016/j.scitotenv.2020.139386>

1379 Bonmatin JM, Moineau I, Charvet R, Colin ME, Fleche C, Bengsch ER (2005a) Behaviour of imidacloprid in
1380 fields. Toxicity for honey bees. In: Lichtfouse E, Schwarzbauer J, Robert D (eds). *Environmental chemistry*.
1381 Springer, Berlin. pp. 483–494. http://link.springer.com/chapter/10.1007%2F3-540-26531-7_44.

1382 Bonmatin JM, Marchand PA, Charvet R, Moineau I, Bengsch ER, Colin ME (2005b) Quantification of
1383 imidacloprid uptake in maize crops. *J Agric Food Chem* 53(13):5336-5341.
1384 <https://doi.org/10.1021/jf0479362>

1385 Bonmatin JM, Giorio C, Girolami V, Goulson D, Kreuzweiser DP, Krupke C, Liess M, Long E, Marzaro M,
1386 Mitchell EAD, Noome DA, Simon-Delso N, Tapparo A (2015) Environmental fate and exposure,
1387 neonicotinoids and fipronil. *Environ Sci Pollut Res* 22(1):35-67. <https://doi.org/10.1007/s11356-014-3332-7>

1388 Botias C, David A, Horwood J, Abdul-Sada A, Nicholls E, Hill E, Goulson D (2015) Neonicotinoid residues in
1389 wildflowers, a potential route of chronic exposure for bees. *Environ Sci Technol* 49(21):12731-12740.
1390 <https://doi.org/10.1021/acs.est.5b03459>

1391 Botias C, David A, Hill EM, Goulson D (2016) Contamination of wild plants near neonicotinoid seed-treated
1392 crops, and implications for non-target insects. *Sci Total Environ* 566:269-278.
1393 <http://dx.doi.org/10.1016/j.scitotenv.2016.05.065>

1394 Botina LL, Bernardes RC, Barbosa WF, Lima MAP, Guedes RNC, Martins GF (2020) Toxicological assessments
1395 of agrochemical effects on stingless bees (Apidae, Meliponini). *MethodsX* 7:100906.
1396 <http://dx.doi.org/10.1016/j.mex.2020.100906>

1397 Bradford BR, Whidden E, Gervasio ED, Checchi PM, Raley-Susman KM (2020) Neonicotinoid-containing
1398 insecticide disruption of growth, locomotion, and fertility in *Caenorhabditis elegans*. *Plos One* 15:e0238637.
1399 <https://doi.org/10.1371/journal.pone.0238637>

1400 Bredeson MM, Lundgren JG (2018) Thiamethoxam seed treatments reduce foliar predator and pollinator
1401 populations in sunflowers (*Helianthus annuus*), and extra-floral nectaries as a route of exposure for seed
1402 treatments to affect the predator, *Coleomegilla maculata* (Coleoptera: Coccinellidae). *Crop Protec* 106:86-
1403 92. <http://dx.doi.org/10.1016/j.cropro.2017.12.019>

1404 Bredeson MM, Reese RN, Lundgren JG (2015) The effects of insecticide dose and herbivore density on tri-trophic
1405 effects of thiamethoxam in a system involving wheat, aphids, and ladybeetles. *Crop Protec* 69:70-76.
1406 <http://dx.doi.org/10.1016/j.cropro.2014.12.010>

1407 Bro E, Decors A, Millot F, Soyeux D, Moinet M, Berny P, Mastain O (2010) Intoxications des perdrix grises en
1408 nature. *Nouveau bilan de la surveillance SAGIR. Faune Sauvage* 289:26-32.

1409 Bro E, Devillers J, Millot F, Decors A (2016) Residues of plant protection products in grey partridge eggs in
1410 French cereal ecosystems. *Environ Sci Pollut Res* 23(10):9559-9573. [http://dx.doi.org/10.1007/s11356-016-](http://dx.doi.org/10.1007/s11356-016-6093-7)
1411 [6093-7](http://dx.doi.org/10.1007/s11356-016-6093-7)

1412 Brodeur JC, Damonte MJ, Rojas DE, Cristos D, Vargas C, Poliserpi MB, Andriulo AE (2022) Concentration of
1413 current-use pesticides in frogs from the Pampa region and correlation of a mixture toxicity index with
1414 biological effects. *Environ Res* 204:112354. <https://doi.org/10.1016/j.envres.2021.112354>

1415 Buchweitz JP, Viner TC, Lehner AF (2019) Qualitative identification of imidacloprid in postmortem animal tissue
1416 by gas chromatography-tandem mass spectrometry. *Toxicol Mech Meth* 29(7):511-517.
1417 <http://dx.doi.org/10.1080/15376516.2019.1616344>

1418 Byholm P, Makelainen S, Santangeli A, Goulson D (2018) First evidence of neonicotinoid residues in a long-
1419 distance migratory raptor, the European honey buzzard (*Pernis apivorus*). *Sci Total Environ* 639:929-933.
1420 <http://dx.doi.org/10.1016/j.scitotenv.2018.05.185>

1421 Camp AA, Batres MA, Williams WC, Koethe RW, Stoner KA, Lehmann DM (2020) Effects of the neonicotinoid
1422 acetamiprid in pollen on *Bombus impatiens* microcolony development. *Environ Tox Chem* 39(12):2560-
1423 2569. <http://dx.doi.org/10.1002/etc.4886>

1424 Camp AA, Lehmann DM (2021) Impacts of neonicotinoids on the bumble bees *Bombus terrestris* and *Bombus*
1425 *impatiens* examined through the lens of an Adverse Outcome Pathway framework. *Environ Tox Chem*
1426 40(2):309-322. <http://dx.doi.org/10.1002/etc.4939>

1427 Campbell KS, Keller PG, Heinzl LM, Golovko SA, Seeger DR, Golovko MY, Kerby JL (2022) Detection of
1428 imidacloprid and metabolites in Northern Leopard frog (*Rana pipiens*) brains. *Sci Total Environ* 813:152424.
1429 <http://dx.doi.org/10.1016/j.scitotenv.2021.152424>

1430 Campbell KS, Keller P, Golovko SA, Seeger D, Golovko MY, Kerby JL (2023) Connecting the pipes: Agricultural
1431 tile drains and elevated imidacloprid brain concentrations in juvenile Northern Leopard frogs (*Rana pipiens*).
1432 *Environ Sci Technol* 57:2758-2767. <https://doi.org/10.1021/acs.est.2c06527>

1433 Cavallaro MC, Liber K, Headley JV, Peru KM, Morrissey CA (2018) Community-level and phenological
1434 responses of emerging aquatic insects exposed to 3 neonicotinoid insecticides: An in situ wetland limnocorral
1435 approach. *Environ Tox Chem* 37(9):2401-2412. <http://dx.doi.org/10.1002/etc.4187>

1436 Cavallaro MC, Main AR, Liber K, Phillips LD, Headley JV, Peru KM, Morrissey CA (2019) Neonicotinoids and
1437 other agricultural stressors collectively modify aquatic insect communities. *Chemosphere* 226:945-955.
1438 <http://dx.doi.org/10.1016/j.chemosphere.2019.03.176>

1439 Chiaia-Hernandez AC, Keller A, Wächter D, Steinlin C, Camenzuli L, Hollender J, Krauss M (2017) Long-term
1440 persistence of pesticides and TPs in archived agricultural soil samples and comparison with pesticide
1441 application. *Environ Sci Technol* 51:10642-10651. <http://dx.doi.org/10.1021/acs.est.7b02529>

- 1442 Chan DSW, Raine NE (2021) Population decline in a ground-nesting solitary squash bee (*Eucera pruinosa*)
1443 following exposure to a neonicotinoid insecticide treated crop (*Cucurbita pepo*). *Scientif Rep* 11:4241.
1444 <http://dx.doi.org/10.1038/s41598-021-83341-7>
- 1445 Chara-Serna AM, Epele LB, Morrissey CA, Richardson JS (2019) Nutrients and sediment modify the impacts of
1446 a neonicotinoid insecticide on freshwater community structure and ecosystem functioning. *Sci Total Environ*
1447 692:1291-1303. <http://dx.doi.org/10.1016/j.scitotenv.2019.06.301>
- 1448 Chauzat MP, Faucon JP, Martel AC, Lachaize J, Cougoule N, Aubert M (2006) A survey of pesticide residues in
1449 pollen loads collected by honey bees in France. *J Econ Entomol* 99(2):253-262.
- 1450 Chauzat MP, Martel AC, Cougoule N, Porta P, Lachaize J, Zeggane S, Aubert M, Carpentier P, Faucon JP (2011)
1451 An assessment of honeybee colony matrices, *apis mellifera* (hymenoptera apidae) to monitor pesticide
1452 presence in continental France. *Environ Tox Chem* 30(1):103-111. <https://doi.org/10.1002/etc.361>
- 1453 Chen Y, Zheng X, Liu J, Wei H, Chen YD, Su XX, Zhang J (2016) Appraisal of the impact of three insecticides
1454 on the principal rice pests and their predators in China. *Florida Entomol* 99(2):210-220.
1455 <http://dx.doi.org/10.1653/024.099.0209>
- 1456 Chen Y, Zhang L, Hu H, Wu R, Ling J, Yue S, Yang D, Yu W, Du W, Shen G, Zhao M (2022) Neonicotinoid
1457 pollution in marine sediments of the East China Sea. *Sci Total Environ* 842:156658.
1458 <https://doi.org/10.1016/j.scitotenv.2022.156658>
- 1459 Coscollà C, Yusà V (2016) Chapter 17 - Pesticides and agricultural air quality. In: de la Guardia, M., Armenta, S.,
1460 eds. *Comprehensive Analytical Chemistry*, Elsevier, 423-490.
1461 <https://www.sciencedirect.com/science/article/pii/S0166526X16300654>
- 1462 Crayton SM, Wood PB, Brown DJ, Millikin AR, McManus TJ, Simpson TJ, Ku KM, Park YL (2020)
1463 Bioaccumulation of the pesticide imidacloprid in stream organisms and sublethal effects on salamanders.
1464 *Glob Ecol Conserv* 24:e01292. <https://doi.org/10.1016/j.gecco.2020.e01292>
- 1465 Criquet J, Dumoulin D, Howsam M, Mondamert L, Goossens JF, Prygiel J, Billon G (2017) Comparison of POCIS
1466 passive samplers vs. composite water sampling: A case study. *Sci Total Environ* 609:982-991.
1467 <https://doi.org/10.1016/j.scitotenv.2017.07.227>
- 1468 Cycoń M, Piotrowska-Seget Z, (2015a) Biochemical and microbial soil functioning after application of the
1469 insecticide imidacloprid. *J Environ Sci* 27:147-158. <http://dx.doi.org/10.1016/j.jes.2014.05.034>

1470 Cycoń M, Piotrowska-Seget Z (2015b) Community structure of ammonia-oxidizing archaea and ammonia-
1471 oxidizing bacteria in soil treated with the insecticide imidacloprid. *Biomed Res Int* 2015:582938.
1472 <http://dx.doi.org/10.1155/2015/582938>

1473 Davis JM, Rosemond AD, Eggert SL, Cross WF, Wallace JB (2010) Long-term nutrient enrichment decouples
1474 predator and prey production. *Proc Natl Acad Sci USA*, 107(1):121-126.
1475 <http://dx.doi.org/10.1073/pnas.0908497107>

1476 de Araújo EP, Dutra Caldas E, Oliveira-Filho EC (2022) Pesticides in surface freshwater: a critical review. *Environ*
1477 *Monit Assess* 194:452. <http://dx.doi.org/10.1007/s10661-022-10005-y>

1478 de Snoo GR, Scheidegger NMI, de Jong FMW (1999) Vertebrate wildlife incidents with pesticides: a European
1479 survey. *Pestic Sci* 55(1):47-54. [https://doi.org/10.1002/\(sici\)1096-9063\(199901\)55:1<47::aid-
1480 ps859>3.3.co;2-r](https://doi.org/10.1002/(sici)1096-9063(199901)55:1<47::aid-ps859>3.3.co;2-r)

1481 Demortain D (2021) The science behind the ban: the outstanding impact of ecotoxicological research on the
1482 regulation of neonicotinoids. *Curr Opin Insect Sci* 46:78-82. <http://dx.doi.org/10.1016/j.cois.2021.02.017>

1483 Désert M, Ravier S, Gille G, Quinapallo A, Armengaud A, Pochet G, Savelli JL, Wortham H, Quivet E (2018)
1484 Spatial and temporal distribution of current-use pesticides in ambient air of Provence-Alpes-Côte-d'Azur
1485 Region and Corsica, France. *Atm Environ* 192:241-256. <https://doi.org/10.1016/j.atmosenv.2018.08.054>

1486 Disque HH, Hamby KA, Dubey A, Taylor C, Dively GP (2019) Effects of clothianidin-treated seed on the
1487 arthropod community in a mid-Atlantic no-till corn agroecosystem. *Pest Manage Sci* 75 (4):969-978.
1488 <http://dx.doi.org/10.1002/ps.5201>

1489 Distefano GG, Zangrando R, Basso M, Panzarin L, Gambaro A, Volpi Ghirardini A, Picone M (2022) The ubiquity
1490 of neonicotinoid contamination: Residues in seabirds with different trophic habits. *Environ Res* 206:112637.
1491 <https://doi.org/10.1016/j.envres.2021.112637>

1492 Dittbrenner N, Triebkorn R, Moser I, Capowiez Y (2010) Physiological and behavioural effects of imidacloprid
1493 on two ecologically relevant earthworm species (*Lumbricus terrestris* and *Aporrectodea caliginosa*).
1494 *Ecotoxicology* 19:1567-1573.

1495 Dittbrenner N, Moser I, Triebkorn R, Capowiez Y (2011a) Assessment of short and long-term effects of
1496 imidacloprid on the burrowing behaviour of two earthworm species (*Aporrectodea caliginosa* and *Lumbricus*
1497 *terrestris*) by using 2D and 3D post-exposure techniques. *Chemosphere* 84:1349-1355.

1498 Dittbrenner N, Schmitt H, Capowiez Y, Triebkorn R (2011b) Sensitivity of *Eisenia fetida* in comparison to
1499 *Aporrectodea caliginosa* and *Lumbricus terrestris* after imidacloprid exposure. Body mass change and
1500 histopathology. *J Soils Sed* 11:1000-1010.

1501 Douglas MR, Rohr JR, Tooker JF (2015) Neonicotinoid insecticide travels through a soil food chain, disrupting
1502 biological control of non-target pests and decreasing soya bean yield. *J Appl Ecol* 52(1):250-260.
1503 <http://dx.doi.org/10.1111/1365-2664.12372>

1504 Douglas MR, Tooker JF (2016) Meta-analysis reveals that seed-applied neonicotinoids and pyrethroids have
1505 similar negative effects on abundance of arthropod natural enemies. *Peerj* 4:e2776.
1506 <http://dx.doi.org/10.7717/peerj.2776>

1507 Dubey A, Lewis MT, Dively GP, Hamby KA (2020) Ecological impacts of pesticide seed treatments on arthropod
1508 communities in a grain crop rotation. *J Appl Ecol* 57(5):936-951. <http://dx.doi.org/10.1111/1365-2664.13595>

1509 Duso C, Ahmad S, Tirello P, Pozzebon A, Klaric V, Baldessari M, Malagnini V, Angeli G (2014) The impact of
1510 insecticides applied in apple orchards on the predatory mite *Kampimodromus aberrans* (Acari: Phytoseiidae).
1511 *Exp Appl Acarol* 62(3):391-414. <http://dx.doi.org/10.1007/s10493-013-9741-3>

1512 EFSA (2018) <https://www.efsa.europa.eu/en/press/news/180228>. Accessed 20 September 2023

1513 EFSA PPR Panel (2015) Statement on the suitability of the BEEHAVE model for its potential use in a regulatory
1514 context and for the risk assessment of multiple stressors in honeybees at the landscape level. *EFSA J* 13:4125.
1515 <https://doi.org/10.2903/j.efsa.2015.4125>

1516 EFSA Scientific Committee, More S, Bampidis V, Benford D, Bragard C, Halldorsson T, Hernández-Jerez A,
1517 Bennekou SH, Koutsoumanis K, Machera K, Naegeli H, Nielsen SS, Schlatter J, Schrenk D, Silano V, Turck
1518 D, Younes M, Arnold G, Dorne J-L, Maggiore A, Pagani S, Szentes C, Terry S, Tosi S, Vrbos D, Zamariola
1519 G and Rortais A (2021) Scientific Opinion on a systems-based approach to the environmental risk assessment
1520 of multiple stressors in honey bees. *EFSA J* 19(5):6607. <https://doi.org/10.2903/j.efsa.2021.6607>

1521 Eng ML, Stutchbury BJM, Morrissey CA (2017) Imidacloprid and chlorpyrifos insecticides impair migratory
1522 ability in a seed-eating songbird. *Scient Rep* 7:15176. <http://dx.doi.org/10.1038/s41598-017-15446-x>

1523 Eng ML, Stutchbury BJM, Morrissey CA (2019) A neonicotinoid insecticide reduces fueling and delays migration
1524 in songbirds. *Science* 365(6458):1177-1180. <http://dx.doi.org/10.1126/science.aaw9419>

1525 English SG, Sandoval-Herrera NI, Bishop CA, Cartwright M, Maisonneuve F, Elliott JE, Welch KC (2021)
1526 Neonicotinoid pesticides exert metabolic effects on avian pollinators. *Scient Rep* 11:2914.
1527 <http://dx.doi.org/10.1038/s41598-021-82470-3>

1528 Ertl HM, Mora MA, Brightsmith DJ, Navarro-Alberto JA (2018) Potential impact of neonicotinoid use on Northern
1529 bobwhite (*Colinus virginianus*) in Texas: A historical analysis. Plos One 13:0191100.
1530 <https://doi.org/10.1371/journal.pone.0191100>

1531 Esquivel CJ, Martinez EJ, Baxter R, Trabanino R, Ranger CM, Michel A, Canas LA (2020) Thiamethoxam
1532 differentially impacts the survival of the generalist predators, *Orius insidiosus* (Hemiptera: Anthocoridae)
1533 and *Hippodamia convergens* (Coleoptera: Coccinellidae), when exposed via the food chain. J Insect Sci
1534 20(4):13, 1-10. <http://dx.doi.org/10.1093/jisesa/ieaa070>

1535 EU Pesticides database (2023) [https://ec.europa.eu/food/plant/pesticides/eu-pesticides-](https://ec.europa.eu/food/plant/pesticides/eu-pesticides-database/start/screen/active-substances)
1536 [database/start/screen/active-substances](https://ec.europa.eu/food/plant/pesticides/eu-pesticides-database/start/screen/active-substances). Accessed 20 September 2023

1537 European Commission (2023) [https://food.ec.europa.eu/plants/pesticides/approval-active-substances/renewal-](https://food.ec.europa.eu/plants/pesticides/approval-active-substances/renewal-approval/neonicotinoids_en)
1538 [approval/neonicotinoids_en](https://food.ec.europa.eu/plants/pesticides/approval-active-substances/renewal-approval/neonicotinoids_en). Accessed 20 September 2023

1539 Evelsizer V, Skopec M (2018) Pesticides, including neonicotinoids, in drained wetlands of Iowa's prairie pothole
1540 region. Wetlands 38(2):221-232. <http://dx.doi.org/10.1007/s13157-016-0796-x>

1541 Faria M, Wu X, Luja-Mondragón M, Prats E, Gómez-Oliván LM, Piña B, Raldúa D (2020) Screening anti-predator
1542 behaviour in fish larvae exposed to environmental pollutants. Sci Total Environ 714:136759.
1543 <https://doi.org/10.1016/j.scitotenv.2020.136759>

1544 Fauser A, Sandrock C, Neumann P, Sadd B (2017) Neonicotinoids override a parasite exposure impact on
1545 hibernation success of a key bumblebee pollinator. Ecol Entomol 42(3):306-314.
1546 <http://dx.doi.org/10.1111/een.12385>

1547 Filimon MN, Voia SO, Popescu R, Dumitrescu G, Ciochina LP, Mituletu M, Vlad DC (2015) The effect of some
1548 insecticides on soil microorganisms based on enzymatic and bacteriological analyses. Rom Biotechnol Lett
1549 20(3):10439-10447.

1550 Fountain MT, Harris AL (2015) Non-target consequences of insecticides used in apple and pear orchards on
1551 *Forficula auricularia* L. (Dermaptera: Forficulidae). Biol Control 91:27-33.
1552 <http://dx.doi.org/10.1016/j.biocontrol.2015.07.007>

1553 Fountain MT, Medd N (2015) Integrating pesticides and predatory mites in soft fruit crops. Phytoparasitica
1554 43(5):657-667. <http://dx.doi.org/10.1007/s12600-015-0485-y>

1555 Frame ST, Pearsons KA, Elkin KR, Saporito LS, Preisendanz HE, Karsten HD, Tooker JF (2021) Assessing
1556 surface and subsurface transport of neonicotinoid insecticides from no-till crop fields. J Environ Qual
1557 50(2):476-484. <http://dx.doi.org/10.1002/jeq2.20185>

1558 Frank SD, Tooker JF (2020) Neonicotinoids pose undocumented threats to food webs. *Proc Natl Acad Sci USA*,
1559 117:22609-22613. <https://doi.org/10.1073/pnas.2017221117>

1560 Fritsch C, Appenzeller B, Burkart L, Coeurdassier M, Scheifler R, Raoul F, Driget V, Powolny T, Gagnaison C,
1561 Rieffel D, Afonso E, Goydadin AC, Hardy EM, Palazzi P, Schaeffer C, Gaba S, Bretagnolle V, Bertrand C,
1562 Pelosi C (2022) Pervasive exposure of wild small mammals to legacy and currently used pesticide mixtures
1563 in arable landscapes. *Sci Rep* 12:15904. <https://doi.org/10.1038/s41598-022-19959-y>

1564 Froger C, Jolivet C, Budzinski H, Pierdet M, Caria G, Saby NPA, Arrouays D, Bispo A (2023). Pesticide residues
1565 in French soils: Occurrence, risks, and persistence. *Environ Sci Technol* 57:7818-7827.
1566 <https://doi.org/10.1021/acs.est.2c09591>

1567 Fuentes E, Gaffard A, Rodrigues A, Millet M, Bretagnolle V, Moreau J, Monceau K (2023) Neonicotinoids: Still
1568 present in farmland birds despite their ban. *Chemosphere* 321:138091.
1569 <https://doi.org/10.1016/j.chemosphere.2023.138091>

1570 Gamble NE, Huff Hartz KE, Figuero AE, Poynton HC, Lydy MJ (2023) Development of insecticide resistance in
1571 *Hyalella azteca*. *Environ Pollut* 322:121165. <https://doi.org/10.1016/j.envpol.2023.121165>

1572 Gibbons D, Morrissey CA, Mineau P (2015) A review of the direct and indirect effects of neonicotinoids and
1573 fipronil on vertebrate wildlife. *Environ Sci Pollut Res* 22(1):103-118. [http://dx.doi.org/10.1007/s11356-014-](http://dx.doi.org/10.1007/s11356-014-3180-5)
1574 3180-5

1575 Gilburn AS, Bunnefeld N, McVean Wilson J, Botham MS, Brereton TM, Fox R, Goulson D (2015) Are
1576 neonicotinoid insecticides driving declines of widespread butterflies? *PeerJ* 3:e1402.
1577 <https://doi.org/10.7717/peerj.1402>

1578 Gill RJ, Raine NE (2014) Chronic impairment of bumblebee natural foraging behaviour induced by sublethal
1579 pesticide exposure. *Funct Ecol* 28(6):1459-1471. <http://dx.doi.org/10.1111/1365-2435.12292>

1580 GIP Loire Bretagne (2013) Micropolluants : Les Pesticides. Cahiers Indicateurs N°1.
1581 [https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&ved=2ahUKEwikgNPv1d3-](https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&ved=2ahUKEwikgNPv1d3-AhUESaQEhb-QAEgQFnoECBkQAQ&url=https%3A%2F%2Fwww.loire-estuaire.org%2Fupload%2Fiedit%2F1%2Fpj%2F51858_2874_L2A6a_micropolluants_pesticides.pdf&usg=AOvVaw3x7V9PEzwdEFrnfyUDo_GT)
1582 [AhUESaQEhb-QAEgQFnoECBkQAQ&url=https%3A%2F%2Fwww.loire-](https://www.loire-estuaire.org%2Fupload%2Fiedit%2F1%2Fpj%2F51858_2874_L2A6a_micropolluants_pesticides.pdf&usg=AOvVaw3x7V9PEzwdEFrnfyUDo_GT)
1583 [estuaire.org%2Fupload%2Fiedit%2F1%2Fpj%2F51858_2874_L2A6a_micropolluants_pesticides.pdf&usg](https://www.loire-estuaire.org%2Fupload%2Fiedit%2F1%2Fpj%2F51858_2874_L2A6a_micropolluants_pesticides.pdf&usg=AOvVaw3x7V9PEzwdEFrnfyUDo_GT)
1584 [=AOvVaw3x7V9PEzwdEFrnfyUDo_GT](https://www.loire-estuaire.org%2Fupload%2Fiedit%2F1%2Fpj%2F51858_2874_L2A6a_micropolluants_pesticides.pdf&usg=AOvVaw3x7V9PEzwdEFrnfyUDo_GT). Accessed 20 September 2023

1585 Gökçe S, Saçan MT (2019) Assessments of algal toxicity and PBT behaviour of pesticides with no eco-
1586 toxicological data: predictive ability of QSA/(T)R models. *Molec Inf* 38:1800137.
1587 <https://doi.org/10.1002/minf.201800137>

1588 Goulson D (2013) An overview of the environmental risks posed by neonicotinoid insecticides. *J Appl Ecol*
1589 50:977-987. [https://doi.org/ 10.1111/1365-2664.12111](https://doi.org/10.1111/1365-2664.12111)

1590 Grassl J, Holt S, Cremen N, Peso M, Hahne D, Baer B (2018) Synergistic effects of pathogen and pesticide
1591 exposure on honey bee (*Apis mellifera*) survival and immunity. *J Invert Pathol* 159:78-86.
1592 <http://dx.doi.org/10.1016/j.jip.2018.10.005>

1593 Graves EE, Meese RJ, Holyoak M (2022) Neonicotinoid exposure in Tricolored Blackbirds (*Agelaius tricolor*).
1594 *Environ Sci Pollut Res* 30:15392-15399. <https://doi.org/10.1007/s11356-022-23290-4>

1595 Gunstone T, Cornelisse T, Klein K, Dubey A, Donley N (2021) Pesticides and soil invertebrates: A hazard
1596 assessment. *Front Environ Sci* 9:643847. <https://doi.org/10.3389/fenvs.2021.643847>

1597 Hallmann CA, Foppen RPB, van Turnhout CAM, de Kroon H, Jongejans E (2014) Declines in insectivorous birds
1598 are associated with high neonicotinoid concentrations. *Nature* 511(7509):341-343+.
1599 <http://dx.doi.org/10.1038/nature13531>

1600 Hayasaka D, Korenaga T, Suzuki K, Saito F, Sánchez-Bayo F, Goka K (2012) Cumulative ecological impacts of
1601 two successive annual treatments of imidacloprid and fipronil on aquatic communities of paddy mesocosms.
1602 *Ecotox Environ Safe* 80:355-362. <http://dx.doi.org/10.1016/j.ecoenv.2012.04.004>

1603 Hayes TB, Falso, P, Gallipeau S, Stice M (2010) The cause of global amphibian declines: a developmental
1604 endocrinologist's perspective. *J Exp Biol* 213:921-933. <http://dx.doi.org/10.1242/jeb.040865>

1605 Henry M, Beguin M, Requier F, Rollin O, Odoux JF, Aupinel P, Aptel J, Tchamitchian S, Decourtye A (2012) A
1606 common pesticide decreases foraging success and survival in honey bees. *Science* 336(6079):348-350.
1607 <http://dx.doi.org/10.1126/science.1215039>

1608 Henry M, Bertrand C, Le Feon V, Requier F, Odoux JF, Aupinel P, Bretagnolle V, Decourtye A (2014) Pesticide
1609 risk assessment in free-ranging bees is weather and landscape dependent. *Nature Comm* 5(1):1-8.
1610 <http://dx.doi.org/10.1038/ncomms5359>

1611 Henry M, Cerrutti N, Aupinel P, Decourtye A, Gayrard M, Odoux JF, Pissard A, Ruger C, Bretagnolle V (2015)
1612 Reconciling laboratory and field assessments of neonicotinoid toxicity to honeybees. *Proc Royal Soc B-Biol*
1613 *Sci* 282:20152110. <http://dx.doi.org/10.1098/rspb.2015.2110>

1614 Henry M, Becher MA, Osborne JL, Kennedy PJ, Aupinel P, Bretagnolle V, Brun F, Grimm V, Horn J, Requier F
1615 (2017) Predictive systems models can help elucidate bee declines driven by multiple combined stressors.
1616 *Apidologie* 48(3):328-339. <http://dx.doi.org/10.1007/s13592-016-0476-0>

1617 Hladik ML, Vandever M, Smalling KL (2016) Exposure of native bees foraging in an agricultural landscape to
1618 current-use pesticides. *Sci Total Environ* 542:469-477. <http://dx.doi.org/10.1016/j.scitotenv.2015.10.077>

1619 Hoffmann AA, Willi Y (2008) Detecting genetic responses to environmental change. *Nature Rev Gen* 9(6):421-
1620 432. <http://dx.doi.org/10.1038/nrg2339>

1621 Hooper SE, Amelon SK, Lin CH (2022) Development of an LC-MS/MS method for non-invasive biomonitoring
1622 of neonicotinoid and systemic herbicide pesticide residues in bat hair. *Toxics* 10:73.
1623 <https://doi.org/10.3390/toxics10020073>

1624 Hsiao CJ, Lin CL, Lin TY, Wang SE, Wu CH (2016) Imidacloprid toxicity impairs spatial memory of echolocation
1625 bats through neural apoptosis in hippocampal CA1 and medial entorhinal cortex areas. *Neuroreport*
1626 27(6):462-468. <http://dx.doi.org/10.1097/wnr.0000000000000562>

1627 Humann-Guilleminot S, Binkowski LJ, Jenni L, Hilke G, Glauser G, Helfenstein F (2019a) A nation-wide survey
1628 of neonicotinoid insecticides in agricultural land with implications for agri-environment schemes. *J Appl*
1629 *Ecol* 56:1502-1514. <https://doi.org/10.1111/1365-2664.13392>

1630 Humann-Guilleminot S, Clement S, Desprat J, Binkowski LJ, Glauser G, Helfenstein F (2019b) A large-scale
1631 survey of house sparrows feathers reveals ubiquitous presence of neonicotinoids in farmlands. *Sci Total*
1632 *Environ* 660:1091-1097. <http://dx.doi.org/10.1016/j.scitotenv.2019.01.068>

1633 Humann-Guilleminot S, de Montaigne CT, Sire J, Grunig S, Gning O, Glauser G, Vallat A, Helfenstein F (2019c)
1634 A sublethal dose of the neonicotinoid insecticide acetamiprid reduces sperm density in a songbird. *Environ*
1635 *Res* 177:108589. <http://dx.doi.org/10.1016/j.envres.2019.108589>

1636 Humann-Guilleminot S, Laurent S, Bize P, Roulin A, Glauser G, Helfenstein F (2021) Contamination by
1637 neonicotinoid insecticides in barn owls (*Tyto alba*) and Alpine swifts (*Tachymarptis melba*). *Sci Total*
1638 *Environ* 785:147403. <http://dx.doi.org/10.1016/j.scitotenv.2021.147403>

1639 Iturburu FG, Simoniello MF, Medici S, Panzeri AM, Menone ML (2018) Imidacloprid Causes DNA damage in
1640 fish: Clastogenesis as a mechanism of genotoxicity. *Bull Environ Cont Toxicol* 100:760-764.
1641 <https://doi.org/10.1007/s00128-018-2338-0>

1642 James DG (2019) A neonicotinoid insecticide at a rate found in nectar reduces longevity but not oogenesis in
1643 monarch butterflies, *Danaus plexippus* (L.). (Lepidoptera: Nymphalidae). *Insects* 10:276.
1644 <http://dx.doi.org/10.3390/insects10090276>

1645 Jones A, Harrington P, Turnbull G (2014) Neonicotinoid concentrations in arable soils after seed treatment
1646 applications in preceding years. *Pest Manage Sci* 70:1780-1784. <https://doi.org/10.1002/ps.3836>

1647 JORF (2021) Arrêté du 5 février 2021 autorisant provisoirement l'emploi de semences de betteraves sucrières
1648 traitées avec des produits phytopharmaceutiques contenant les substances actives imidaclopride ou
1649 thiamethoxam. Journal Officiel de la République Française, Texte 25 sur 113, 6 février 2021.
1650 <https://www.legifrance.gouv.fr/download/pdf?id=hbs9mp5->
1651 [XF68jgbjPxIB6NVeMJTi2C06ic6UTBmB0po=](https://www.legifrance.gouv.fr/download/pdf?id=hbs9mp5-XF68jgbjPxIB6NVeMJTi2C06ic6UTBmB0po=). Accessed 20 September 2023

1652 JORF (2022) Arrêté du 31 janvier 2022 autorisant provisoirement l'emploi de semences de betteraves sucrières
1653 traitées avec des produits phytopharmaceutiques contenant les substances actives imidaclopride ou
1654 thiamethoxam et précisant les cultures qui peuvent être semées, plantées ou replantées au titre des campagnes
1655 suivantes. Journal Officiel de la République Française, Texte 44 sur 145, 1^{er} février 2022.
1656 <https://www.legifrance.gouv.fr/download/pdf?id=6WRjAfdMSHV4TefP9->
1657 [NdUXcMTd9GOIfKK9zcag5Ik0E=](https://www.legifrance.gouv.fr/download/pdf?id=6WRjAfdMSHV4TefP9-NdUXcMTd9GOIfKK9zcag5Ik0E=). Accessed 20 September 2023

1658 Kaczynski P, Lozowicka B, Perkowski M, Zon W, Hrynkó I, Rutkowska E, Skibko Z (2021) Impact of broad-
1659 spectrum pesticides used in the agricultural and forestry sector on the pesticide profile in wild boar, roe deer
1660 and deer and risk assessment for venison consumers. *Sci Total Environ* 784:147215.
1661 <http://dx.doi.org/10.1016/j.scitotenv.2021.147215>

1662 Kattwinkel M, Reichert P, Ruegg J, Liess M, Schuwirth N (2016) Modeling macroinvertebrate community
1663 dynamics in stream mesocosms contaminated with a pesticide. *Environ Sci Technol* 50(6):3165-3173.
1664 <http://dx.doi.org/10.1021/acs.est.5b04068>

1665 Kenna D, Cooley H, Pretelli I, Rodrigues AR, Gill SD, Gill RJ (2019) Pesticide exposure affects flight dynamics
1666 and reduces flight endurance in bumblebees. *Ecol Evol* 9(10):5637-5650.
1667 <http://dx.doi.org/10.1002/ece3.5143>

1668 Khans FR, Alhewairini SS (2019) Effects of insecticides on natural population of hymenopterous parasitoids in
1669 Alfalfa (*Medicago sativa* L.) agro-ecosystem. *Pakistan J Agric Sci* 56(4):1087-1093.

1670 Khidkhan K, Poapolathep A, Poapolathep S (2022) Residues of neonicotinoids and fipronil in paddy fields and
1671 duck eggs: Do ducks transform and accumulate these substances to egg. *J Toxicol Sci* P-22E.
1672 https://doi.org/10.14869/toxpt.49.1.0_P-22E

1673 Kimura-Kuroda J, Komuta Y, Kuroda Y, Hayashi M, Kawano H (2012) Nicotine-like effects of the neonicotinoid
1674 insecticides acetamiprid and imidacloprid on cerebellar neurons from neonatal rats. *Plos One* 7:e32432.
1675 <https://doi.org/10.1371/journal.pone.0032432>

1676 Klaus F, Tschardt T, Bischoff G, Grass I (2021) Floral resource diversification promotes solitary bee
1677 reproduction and may offset insecticide effects - evidence from a semi-field experiment. *Ecol Lett* 24(4):668-
1678 675. <http://dx.doi.org/10.1111/ele.13683>

1679 Klingelhöfer D, Braun M, Brüggman D, Groneberg DA (2022) Neonicotinoids: A critical assessment of the global
1680 research landscape of the most extensively used insecticide. *Environ Res* 213:113727.
1681 <https://doi.org/10.1016/j.envres.2022.113727>

1682 Kobashi K, Harada T, Adachi Y, Mori M, Ihara M, Hayasaka D (2017) Comparative ecotoxicity of imidacloprid
1683 and dinotefuran to aquatic insects in rice mesocosms. *Ecotox Environ Safe* 138:122-129.
1684 <https://doi.org/10.1016/j.ecoenv.2016.12.025>

1685 Könemann S, Meyer S, Betz A, Županič A, vom Berg C (2021) Sub-Lethal Peak Exposure to Insecticides Triggers
1686 Olfaction-Mediated Avoidance in Zebrafish Larvae. *Environ Sci Technol* 55:11835-11847.
1687 <https://doi.org/10.1021/acs.est.1c01792>

1688 Koppenhöfer AM, Cowles RS, Cowles EA, Fuzy EM, Kaya HK (2003) Effect of neonicotinoid synergists on
1689 entomopathogenic nematode fitness. *Entomol Exp Applic* 106(1):7-18. <http://dx.doi.org/10.1046/j.1570->
1690 [7458.2003.00008.x](http://dx.doi.org/10.1046/j.1570-7458.2003.00008.x)

1691 Kraus JM, Kuivila KM, Hladik ML, Shook N, Mushet DM, Dowdy K, Harrington R (2021) Cross-ecosystem
1692 fluxes of pesticides from prairie wetlands mediated by aquatic insect emergence: Implications for terrestrial
1693 insectivores. *Environ Tox Chem* 40(8):2282-2296. <http://dx.doi.org/10.1002/etc.5111>

1694 Kreutzweiser D, Good K, Chartrand D, Scarr T, Thompson D (2007) Non-target effects on aquatic decomposer
1695 organisms of imidacloprid as a systemic insecticide to control emerald ash borer in riparian trees. *Ecotox*
1696 *Environ Safe* 68(3):315-325. <http://dx.doi.org/10.1016/j.ecoenv.2007.04.011>

1697 Kreutzweiser DP, Good KP, Chartrand DT, Scarr TA, Thompson DG (2008) Toxicity of the systemic insecticide,
1698 imidacloprid, to forest stream insects and microbial communities. *Bull Environ Contam Toxicol* 80(3):211-
1699 214. <http://dx.doi.org/10.1007/s00128-007-9347-8>

1700 Krupke CH, Hunt GJ, Eitzer BD, Andino G, Given K (2012) Multiple routes of pesticide exposure for honey bees
1701 living near agricultural fields. *Plos One* 7(1):e29268. <http://dx.doi.org/10.1371/journal.pone.0029268>

1702 Krupke CH, Long EY (2015) Intersections between neonicotinoid seed treatments and honey bees. *Curr Op Insect*
1703 *Sci* 10:8-13. <http://dx.doi.org/10.1016/j.cois.2015.04.005>

1704 Kudelska MM, Holden-Dye L, O'Connor V, Doyle DA (2017) Concentration-dependent effects of acute and
1705 chronic neonicotinoid exposure on the behaviour and development of the nematode *Caenorhabditis elegans*.
1706 *Pest Manage Sci* 73:1345-1351. <http://dx.doi.org/10.1002/ps.4564>

1707 Kuechle KJ, Webb EB, Mengel D, Main AR (2019) Factors influencing neonicotinoid insecticide concentrations
1708 in floodplain wetland sediments across Missouri. *Environ Sci Technol* 53(18):10591-10600.
1709 <http://dx.doi.org/10.1021/acs.est.9b01799>

1710 Kuzukiran O, Simsek I, Yorulmaz T, Yurdakok-Dikmen B, Ozkan O, Filazi A (2021) Multiresidues of
1711 environmental contaminants in bats from Turkey. *Chemosphere* 282:131022.
1712 <https://doi.org/10.1016/j.chemosphere.2021.131022>

1713 LaLone CA, Villeneuve DL, Wu-Smart J, Milsk RY, Sappington K, Garber KV, Housenger J, Ankley GT (2017)
1714 Weight of evidence evaluation of a network of adverse outcome pathways linking activation of the nicotinic
1715 acetylcholine receptor in honey bees to colony death. *Sci Total Environ* 584:751-775.
1716 <http://dx.doi.org/10.1016/j.scitotenv.2017.01.113>

1717 Laycock I, Lenthall KM, Barratt AT, Cresswell JE (2012) Effects of imidacloprid, a neonicotinoid pesticide, on
1718 reproduction in worker bumble bees (*Bombus terrestris*). *Ecotoxicology* 21(7):1937-1945.
1719 <http://dx.doi.org/10.1007/s10646-012-0927-y>

1720 Lee-Jenkins SSY, Robinson SA (2018) Effects of neonicotinoids on putative escape behavior of juvenile wood
1721 frogs (*Lithobates sylvaticus*) chronically exposed as tadpoles. *Environ Tox Chem* 37(12):3115-3123.
1722 <https://doi.org/10.1002/etc.4284>

1723 Lennon RJ, Isaac NJB, Shore RF, Peach WJ, Dunn JC, Pereira MG, Arnold KE, Garthwaite D, Brown CD (2019)
1724 Using long-term datasets to assess the impacts of dietary exposure to neonicotinoids on farmland bird
1725 populations in England. *Plos One* 14(10):1-21. <http://dx.doi.org/10.1371/journal.pone.0223093>

1726 Lennon RJ, Shore RF, Pereira MG, Peach WJ, Dunn JC, Arnold KE, Brown CD (2020a) High prevalence of the
1727 neonicotinoid clothianidin in liver and plasma samples collected from gamebirds during autumn sowing. *Sci*
1728 *Total Environ* 742:140493. <http://dx.doi.org/10.1016/j.scitotenv.2020.140493>

1729 Lennon RJ, Peach WJ, Dunn JC, Shore RF, Pereira MG, Sleep D, Dodd S, Wheatley CJ, Arnold KE, Brown CD
1730 (2020b) From seeds to plasma: Confirmed exposure of multiple farmland bird species to clothianidin during
1731 sowing of winter cereals. *Sci Total Environ* 723:138056. <http://dx.doi.org/10.1016/j.scitotenv.2020.138056>

1732 Levesque B, Cachot J, Boet P, Lepage M, Mazella N, Martin C, Gourves PY, Legeay A (2018) Seasonal variations
1733 of contamination and exoskeletal malformations in the white shrimps *Palaemon longirostris* in the Gironde
1734 estuary, France. *Environ Sci Pollut Res* 25(23):22689-22701. <https://doi.org/10.1007/s11356-018-2241-6>

1735 Li L, Chen X, Zhang D, Pan X (2010) Effects of insecticide acetamiprid on Photosystem II (PSII) activity of
1736 *Synechocystis* sp. (FACHB-898). *Pestic Biochem Physiol* 98:300-304.
1737 <https://doi.org/10.1016/j.pestbp.2010.06.022>

1738 Li MD, Yang Z, Guo H, Dash B (2016) Evolutionary relationship of nicotinic acetylcholine receptor subunits in
1739 both vertebrate and invertebrate species. *Neuromethods* 117:227-254. [https://doi.org/10.1007/978-1-4939-](https://doi.org/10.1007/978-1-4939-3768-4_12)
1740 [3768-4_12](https://doi.org/10.1007/978-1-4939-3768-4_12)

1741 Li YF, An JJ, Dang ZH, Lv HY, Pan WL, Gao ZL (2018) Treating wheat seeds with neonicotinoid insecticides
1742 does not harm the rhizosphere microbial community. *Plos One* 13(12):1-12.
1743 <http://dx.doi.org/10.1371/journal.pone.0205200>

1744 Li YJ, Miao RQ, Khanna M (2020) Neonicotinoids and decline in bird biodiversity in the United States. *Nature*
1745 *Sustain* 3:1027-1035. <http://dx.doi.org/10.1038/s41893-020-0582-x>

1746 Lima-Fernandes E, Bundschuh M, Bakanov N, Englert D, Schulz R, Schafer RB (2019) Effects of a systemic
1747 pesticide along an aquatic tri-trophic food chain. *Bull Environ Contam Toxicol* 103:507-514.
1748 <http://dx.doi.org/10.1007/s00128-019-02696-w>

1749 Lin CH, Sponsler DB, Richardson RT, Watters HD, Glinski DA, Henderson WM, Minucci JM, Lee EH, Purucker
1750 ST, Johnson RM (2021) Honey bees and neonicotinoid-treated corn seed: Contamination, exposure, and
1751 effects. *Environ Tox Chem* 40(4):1212-1221. <http://dx.doi.org/10.1002/etc.4957>

1752 Liu H, Tang X, Xu X, Dai Y, Zhang X, Yang Y (2021) Potential for phytoremediation of neonicotinoids by nine
1753 wetland plants. *Chemosphere* 283:131083. <https://doi.org/10.1016/j.chemosphere.2021.131083>

1754 Lopez-Antia A, Ortiz-Santaliestra ME, Mougeot F, Mateo R (2013) Experimental exposure of red-legged
1755 partridges (*Alectoris rufa*) to seeds coated with imidacloprid, thiram and difenoconazole. *Ecotoxicology*
1756 22(1):125-138. <http://dx.doi.org/10.1007/s10646-012-1009-x>

1757 Lopez-Antia A, Ortiz-Santaliestra ME, Mateo R (2014) Experimental approaches to test pesticide-treated seed
1758 avoidance by birds under a simulated diversification of food sources. *Sci Total Environ*, 496: 179-187.
1759 <http://dx.doi.org/10.1016/j.scitotenv.2014.07.031>

1760 Lopez-Antia A, Ortiz-Santaliestra ME, Mougeot F, Mateo R (2015) Imidacloprid-treated seed ingestion has lethal
1761 effect on adult partridges and reduces both breeding investment and offspring immunity. *Environ Res* 136:97-
1762 107. <http://dx.doi.org/10.1016/j.envres.2014.10.023>

1763 Lopez-Antia A, Feliu J, Camarero PR, Ortiz Santaliestra ME, Mateo R (2016) Risk assessment of pesticide seed
1764 treatment for farmland birds using refined field data. *J Appl Ecol* 53(5):1373-1381.
1765 <https://doi.org/10.1111/1365-2664.12668>

1766 Lundin O, Rundlöf M, Smith HG, Fries I, Bommarco R (2015) Neonicotinoid insecticides and their impacts on
1767 bees: A systematic review of research approaches and identification of knowledge gaps. *Plos One*
1768 10(8):e0136928. <http://dx.doi.org/10.1371/journal.pone.0136928>

1769 MacDonald AM, Jardine CM, Thomas PJ, Nemeth NM (2018) Neonicotinoid detection in wild turkeys (*Meleagris*
1770 *gallopavo silvestris*) in Ontario, Canada. *Environ Sci Pollut Res* 25(16):16254-16260.
1771 <http://dx.doi.org/10.1007/s11356-018-2093-0>

1772 Main AR, Fehr J, Liber K, Headley JV, Peru KM, Morrissey CA (2017) Reduction of neonicotinoid insecticide
1773 residues in Prairie wetlands by common wetland plants. *Sci Total Environ* 579:1193-1202.
1774 <http://dx.doi.org/10.1016/j.scitotenv.2016.11.102>

1775 Main AR, Webb EB, Goynes KW, Mengel D (2020) Reduced species richness of native bees in field margins
1776 associated with neonicotinoid concentrations in non-target soils. *Agric Ecosys Environ* 287:106693.
1777 <http://dx.doi.org/10.1016/j.agee.2019.106693>

1778 Malagnoux L, Capowiez Y, Rault M (2015) Impact of insecticide exposure on the predation activity of the
1779 European earwig *Forficula auricularia*. *Environ Sci Pollut Res* 22:14116-14126.
1780 <http://dx.doi.org/10.1007/s11356-015-4520-9>

1781 Malev O, Sauerborn Klobučar R, Fabbretti E, Trebse P (2012) Comparative toxicity of imidacloprid and its
1782 transformation product 6-chloronicotinic acid to non-target organisms: microalgae *Desmodesmus*
1783 *subspicatus* and amphipod *Gammarus fossarum*. *Pestic Biochem Physiol* 104:178-186.
1784 <http://dx.doi.org/10.1016/j.pestbp.2012.07.008>

1785 Mansoor MM, Shad SA (2020) Genetics, cross-resistance and realized heritability of resistance to acetamiprid in
1786 generalist predator, *Chrysoperla carnea* (Steph.) (Neuroptera: Chrysopidae). *Egyptian J Biol Pest Cont* 30:23.
1787 <http://dx.doi.org/10.1186/s41938-020-0213-x>

1788 Mburia R (2016) The use of ducks in rice fields in the control of weeds and pests. The Permaculture Research
1789 Institute. <https://www.permaculturenews.org/2016/09/15/use-ducks-rice-fields-control-weed-pest/> Accessed
1790 20 September 2023

1791 McGee S, Whitfield-Aslund M, Duca D, Kopysh N, Dan T, Knopper L, Brewer L (2018) Field evaluation of the
1792 potential for avian exposure to clothianidin following the planting of clothianidin-treated corn seed. PeerJ
1793 6:e5880. <http://dx.doi.org/10.7717/peerj.5880>

1794 Menet-Nedelec F, Gonzalez JL, Halm-Lemeille MP, Repecaud M, Facq JV, Maheux F, Pierre-Duplessix O, Simon
1795 B (2018) Etude d'outils d'évaluation de la contamination chimique dans les eaux de la Manche – ECUME
1796 Rapport Ifremer ODE/UL/LERN/18-02, 53. <http://archimer.ifremer.fr/doc/00434/54582/55966.pdf>

1797 Miles JC, Hua J, Sepulveda MS, Krupke CH, Hoverman JT (2017) Effects of clothianidin on aquatic communities:
1798 Evaluating the impacts of lethal and sublethal exposure to neonicotinoids. PLoS One 12:1-24.
1799 <https://doi.org/10.1371/journal.pone.0174171>

1800 Miller JL, Schmidt TS, van Metre PC, Mahler BJ, Sandstrom MW, Nowell LH, Carlisle DM, Moran PW (2020)
1801 Common insecticide disrupts aquatic communities: A mesocosm-to-field ecological risk assessment of
1802 fipronil and its degradates in U.S. streams. Science Adv 6:1-12. <https://doi.org/10.1126/sciadv.abc1299>

1803 Millot F, Decors A, Mastain O, Quintaine T, Berny P, Vey D, Lasseur R, Bro E (2017) Field evidence of bird
1804 poisonings by imidacloprid-treated seeds: a review of incidents reported by the French SAGIR network from
1805 1995 to 2014. Environ Sci Pollut Res 24(6):5469-5485. <http://dx.doi.org/10.1007/s11356-016-8272-y>

1806 Mineau P (2017) Organophosphorous and carbamate insecticides: Impacts on birds. Encycl Anthrope 111-117.
1807 <http://dx.doi.org/10.1016/B978-0-12-409548-9.09887-0>

1808 Mineau P, Callaghan C (2018) Neonicotinoid insecticides and bats: an assessment of the direct and indirect risks.
1809 Ontario, Canada: Canadian Wildlife Federation, 83 p.

1810 Mineau P, Kern H (2023). Neonicotinoid insecticides: Failing to come to grips with a predictable environmental
1811 disaster. American Bird Conservancy, June 2023, 121 p.

1812 Mineau P, Palmer C (2013) The impact of the nation's most widely used insecticides on birds. American Bird
1813 Conservancy, USA, 2013: American Bird Conservancy, Neonicotinoid Insecticides and Birds, 96 p.

1814 Mitchell EAD, Mulhauser B, Mulot M, Mutabazi A, Glauser G, Aebi A (2017) A worldwide survey of
1815 neonicotinoids in honey. Science 358(6359):109-111. <https://doi.org/10.1126/science.aan3684>

1816 Mohr S, Berghahn R, Schmiediche R, Hübner V, Loth S, Feibicke M, Mailahn W, Wogram J (2012)
1817 Macroinvertebrate community response to repeated short-term pulses of the insecticide imidacloprid. *Aquat*
1818 *Toxicol* 110:25-36. <https://doi.org/10.1016/j.aquatox.2011.11.016>

1819 Monchanin C, Henry M, Decourtye A, Dalmon A, Fortini D, Boeuf E, Dubuisson L, Aupinel P, Chevallereau C,
1820 Petit J, Fourrier J (2019) Hazard of a neonicotinoid insecticide on the homing flight of the honeybee depends
1821 on climatic conditions and *Varroa* infestation. *Chemosphere* 224:360-368.
1822 <http://dx.doi.org/10.1016/j.chemosphere.2019.02.129>

1823 Moreau J, Rabdeau J, Badenhausser I, Giraudeau M, Sepp T, Crépin M, Gaffard A, Bretagnolle V, Monceau K
1824 (2022) Pesticide impacts on avian species with special reference to farmland birds: a review. *Environ Monit*
1825 *Assess* 194:790. <https://doi.org/10.1007/s10661-022-10394-0>

1826 Morrissey CA, Mineau P, Devries JH, Sánchez-Bayo F, Liess M, Cavallaro MC, Liber K (2015) Neonicotinoid
1827 contamination of global surface waters and associated risk to aquatic invertebrates: A review. *Environ Int*
1828 74:291-303. <https://doi.org/10.1016/j.envint.2014.10.024>

1829 Müller C (2018) Impacts of sublethal insecticide exposure on insects - Facts and knowledge gaps. *Basic App Ecol*
1830 30:1-10. <http://dx.doi.org/10.1016/j.baae.2018.05.001>

1831 Munaron D, Derolez V, Foucault E, Cimiterra N, Tapie N, Budzinski H, Giraud A (2020) OBSLAG – Volet
1832 Pesticides. Bilan 2017-2019 du suivi des lagunes méditerranéennes. Rapport Ifremer n° ODE/UL/LER-
1833 LR/20.09. <https://archimer.ifremer.fr/doc/00656/76769/> Accessed 20 September 2023

1834 Munaron D, Gianaroli C, Cimiterra N, Derolez V, Ouisse V, Giraud A (2022) OBSLAG - Pesticides. Bilan 2020-
1835 2021 du suivi des lagunes méditerranéennes. Rapport de la Convention AERMC-Ifremer 2020, n°
1836 ODE/UL/LER-LR/22.12, août 2022. 90p. <https://archimer.ifremer.fr/doc/00795/90672/>. Accessed 20
1837 September 2023

1838 Munaron D, Mérigot B, Derolez V, Tapie N, Budzinski H, Fiandrino A (2023) Evaluating pesticide mixture risks
1839 in French Mediterranean coastal lagoons waters. *Sci Total Environ* 867:161303.
1840 <https://doi.org/10.1016/j.scitotenv.2022.161303>

1841 Nafaji G, Razi M, Hoshyar A, Shahmohamadloo S, Feyzi S (2010) The effect of chronic exposure with
1842 imidacloprid insecticide on fertility in mature male rats. *Int J Fertil Steril* 4:9-16.

1843 Naranjo SE (2001) Conservation and evaluation of natural enemies in IPM systems for *Bemisia tabaci*. *Crop Protec*
1844 20(9):835-852. [http://dx.doi.org/10.1016/s0261-2194\(01\)00115-6](http://dx.doi.org/10.1016/s0261-2194(01)00115-6)

1845 Naumann T, Bento CPM, Wittmann A, Gandrass J, Tang J, Zhen X, Liu L, Ebinghaus R (2022) Occurrence and
1846 ecological risk assessment of neonicotinoids and related insecticides in the Bohai Sea and its surrounding
1847 rivers, China. *Water Res* 209:117912. [https://doi.org/ 10.1016/j.watres.2021.117912](https://doi.org/10.1016/j.watres.2021.117912)

1848 Néliou S, Lamy I, Karolak S, Delarue G, Crouzet O, Barraud C, Bimbot M, Allaoui F, Hanot C, Delorme A, Levi
1849 Y, Hulot FD, Baudry E (2021) Impact of peri-urban landscape on the organic and mineral contamination of
1850 pond waters and related risk assessment. *Environ Sci Pollut Res* 28:59256-59267.
1851 <https://doi.org/10.1007/s11356-020-10355-5>

1852 Neury-Ormanni J, Doose C, Majdi N, Vedrenne J, Morin S, Höss S, Traunspurger W (2019) Tolerance of free-
1853 living nematode species to imidacloprid and diuron. *Invertebr Biol* 138:e12272.
1854 <https://doi.org/10.1111/ivb.12272>

1855 Neury-Ormanni J, Vedrenne J, Morin S (2020a) Benthic diatom growth kinetics under combined pressures of
1856 microalgal competition, predation and chemical stressors. *Sci Total Environ* 734:139484.
1857 <https://doi.org/10.1016/j.scitotenv.2020.139484>

1858 Neury-Ormanni J, Doose C, Majdi N, Vedrenne J, Traunspurger W, Morin S (2020b) Selective grazing behaviour
1859 of chironomids on microalgae under pesticide pressure. *Sci Total Environ* 730:138673.
1860 <http://dx.doi.org/10.1016/j.scitotenv.2020.138673>

1861 Oliveira JM, Destro ALF, Freitas MB, Oliveira LL (2021) How do pesticides affect bats? - A brief review of recent
1862 publications. *Braz J Biol* 81(2):499-507. <http://dx.doi.org/10.1590/1519-6984.225330>

1863 Oziolor EM, De Schampelaere K, Matson CW (2016) Evolutionary toxicology: Meta-analysis of evolutionary
1864 events in response to chemical stressors. *Ecotoxicology* 25(10):1858-1866.
1865 <http://dx.doi.org/10.1007/s10646-016-1735-6>

1866 Pearsons KA, Tooker JF (2021) Preventive insecticide use affects arthropod decomposers and decomposition in
1867 field crops. *App Soil Ecol* 157:103757. <http://dx.doi.org/10.1016/j.apsoil.2020.103757>

1868 Pekar S (2012) Spiders (Araneae) in the pesticide world: an ecotoxicological review. *Pest Manage Sci*
1869 68(11):1438-1446. <http://dx.doi.org/10.1002/ps.3397>

1870 Pekar S, Kocourek F (2004) Spiders (Araneae) in the biological and integrated pest management of apple in the
1871 Czech Republic. *J App Entomol* 128(8):561-566. <http://dx.doi.org/10.1111/j.1439-0418.2004.00884.x>

1872 Pelosi C, Bertrand C, Daniele G, Coeurdassier M, Benoit P, Néliou S, Lafay F, Bretagnolle V, Gaba S, Vulliet E,
1873 Fritsch C (2021) Residues of currently used pesticides in soils and earthworms: A silent threat? *Agric Ecosyst*
1874 *Environ* 305:107167. <http://dx.doi.org/10.1016/j.agee.2020.107167>

1875 Penn HJ, Dale AM (2017) Imidacloprid seed treatments affect individual ant behavior and community structure
1876 but not egg predation, pest abundance or soybean yield. *Pest Manage Sci* 73(8):1625-1632.
1877 <http://dx.doi.org/10.1002/ps.4499>

1878 Pepin JF, Benabdelmouna A, Degremont L, Guesdon S, Le Moine O, Morga B, Bierne N, Travers MA, Robert S,
1879 Soletchnik P (2017) Mortalités de moules bleues dans les secteurs mytilicoles charentais et vendéens :
1880 description et facteurs liés – MORBLEU. Rapport Ifremer RBE/SG2M-LGPMM, 93p.

1881 Pesce S, Mamy L, Achard AL, Le Gall M, Le Perchec S, Réchauchère O, Tibi A, Leenhardt S, Sanchez W (2021)
1882 Collective scientific assessment as a relevant tool to inform public debate and policymaking: an illustration
1883 about the effects of plant protection products on biodiversity and ecosystem services. *Environ Sci Pollut Res*
1884 28:38448-38454. <https://doi.org/10.1007/s11356-021-14863-w>

1885 Pesce S, Mamy L, Sanchez W, Amichot M, Artigas J, Aviron S, Barthélémy C, Beaudouin R, Bedos C, Bérard A,
1886 Berny P, Bertrand C, Bertrand C, Betouille S, Bureau-Point E, Charles S, Chaumot A, Chauvel B,
1887 Coeurdassier M, Corio-Costet MF, Coutellec MA, Crouzet O, Doussan I, Faburé J, Fritsch C, Gallai N,
1888 Gonzalez P, Gouy V, Hedde M, Langlais A, Le Bellec F, Leboulanger C, Margoum C, Martin-Laurent F,
1889 Mongruel R, Morin S, Mougin C, Munaron D, Néliou S, Pelosi C, Rault M, Sabater S, Stachowski-Haberkorn
1890 S, Sucre E, Thomas M, Tournebize J, Leenhardt S (2023) Main conclusions and perspectives from the
1891 collective scientific assessment on the effects of plant protection products on biodiversity and ecosystem
1892 services along the land–sea continuum in France and French overseas territories. *Environ Sci Pollut Res* (in
1893 press). <https://doi.org/10.1007/s11356-023-26952-z>

1894 Pettis JS, Lichtenberg EM, Andree M, Stitzinger J, Rose R, Vanengelsdorp D (2013) Crop pollination exposes
1895 honey bees to pesticides which alters their susceptibility to the gut pathogen *Nosema ceranae*. *Plos One*
1896 8(7):9. <http://dx.doi.org/10.1371/journal.pone.0070182>

1897 Phytatmo database (2023) <https://www.atmo-france.org/article/phytatmo>. Accessed 20 September 2023

1898 Picone M, Distefano GG, Zangrando R, Gambaro A, Volpi Ghirardini A (2023). Neonicotinoids and
1899 pharmaceuticals in hair of the Red fox (*Vulpes vulpes*) from the Cavallino-Treporti peninsula, Italy. *Environ*
1900 *Res* 228:115837. <https://doi.org/10.1016/j.envres.2023.115837>

1901 Pietrzak D, Kania J, Malina G, Kmiecik E, Wator K (2019) Pesticides from the EU first and second watch lists in
1902 the water environment. *Clean-Soil Air Water* 47(7):1800376. <https://doi.org/10.1002/clen.201800376>

1903 Pilling E, Campbell P, Coulson M, Ruddle N, Tornier I (2013) A four-year field program investigating long-term
 1904 effects of repeated exposure of honey bee colonies to flowering crops treated with thiamethoxam. *Plos One*
 1905 8(10):14. <https://doi.org/10.1371/journal.pone.0077193>

1906 Pisa LW, Amaral-Rogers V, Belzunces LP, Bonmatin JM, Downs CA, Goulson D, Kreuzweiser DP, Krupke C,
 1907 Liess M, McField M, Morrissey CA, Noome DA, Settele J, Simon-Delso N, Stark JD, Van der Sluijs JP, Van
 1908 Dyck H, Wiemers M (2015) Effects of neonicotinoids and fipronil on non-target invertebrates. *Environ Sci*
 1909 *Pollut Res* 22(1):68-102. <http://dx.doi.org/10.1007/s11356-014-3471-x>

1910 Poisson MC, Garrett DR, Sigouin A, Belisle M, Garant D, Haroune L, Bellenger JP, Pelletier F (2021) Assessing
 1911 pesticides exposure effects on the reproductive performance of a declining aerial insectivore. *Ecol Applic*
 1912 31(7):e02415. <http://dx.doi.org/10.1002/eap.2415>

1913 Pozzebon A, Duso C, Tirello P, Ortiz PB (2011) Toxicity of thiamethoxam to *Tetranychus urticae* Koch and
 1914 *Phytoseiulus persimilis* Athias-Henriot (Acari Tetranychidae, Phytoseiidae) through different routes of
 1915 exposure. *Pest Manage Sci* 67:352-359. <https://doi.org/10.1002/ps.2072>

1916 PPDB (2023) Pesticide Properties DataBase. <http://sitem.herts.ac.uk/aeru/ppdb/>. Accessed 20 September 2023

1917 Preuss TG, Agatz A, Goussen B, Roeben V, Rumkee J, Zakharova L, Thorbek P (2022) The BEEHAVE_ecotox
 1918 model - Integrating a mechanistic effect module into the honeybee colony model. *Environ Toxicol Chem*
 1919 41:2870-2882. <https://doi.org/10.1002/etc.5467>

1920 Prosser P, Hart ADM (2005) Assessing potential exposure of birds to pesticide-treated seed. *Ecotoxicology*
 1921 14:679-691. <https://doi.org/10.1007/s10646-005-0018-4>

1922 Prouteau L (2021) Caractérisation de la contamination en pesticides azoles et néonicotinoïdes chez les espèces
 1923 d'intérêt localisées en région Nouvelle-Aquitaine : développement de méthodes analytiques et applications
 1924 (PhD Thesis). La Rochelle Université, La Rochelle, France.

1925 Puglis HJ, Boone MD (2011) Effects of technical-grade active ingredient vs. commercial formulation of seven
 1926 pesticides in the presence or absence of UV radiation on survival of green frog tadpoles. *Arch Environ*
 1927 *Contam Toxicol* 60:145-155. <https://doi.org/10.1007/s00244-010-9528-z>

1928 Qi S, Wang D, Zhu L, Teng M, Wang C, Xue X, Wu L (2018) Effects of a novel neonicotinoid insecticide
 1929 cycloxaprid on earthworm, *Eisenia fetida*. *Environ Sci Pollut Res* 25:14138-14147.
 1930 <https://doi.org/10.1007/s11356-018-1624-z>

1931 Raina-Fulton R (2015) Determination of neonicotinoid insecticides and strobilurin fungicides in particle phase
1932 atmospheric samples by liquid chromatography-tandem mass spectrometry. *J Agric Food Chem* 63(21):5152-
1933 5162. <https://doi.org/10.1021/acs.jafc.5b01347>

1934 Resende-Silva GA, Turchen LM, Guedes RNC, Cutler GC (2019) Imidacloprid soil drenches affect weight and
1935 functional response of spined soldier bug (Hemiptera: Pentatomidae). *J Eco Entomol* 112(2):558-564.
1936 <http://dx.doi.org/10.1093/jee/toy401>

1937 Rico A, Arenas-Sánchez A, Pasqualini J, García-Astillero A, Cherta L, Nozal L, Vighi M (2018) Effects of
1938 imidacloprid and a neonicotinoid mixture on aquatic invertebrate communities under Mediterranean
1939 conditions. *Aquat Toxicol* 204:130-143. <http://dx.doi.org/10.1016/j.aquatox.2018.09.004>

1940 Riedo J, Wettstein FE, Rösch A, Herzog C, Banerjee S, Büchi L, Charles R, Wächter D, Martin-Laurent F,
1941 Bucheli TD, Walder T, van der Heijden MGA (2021) Widespread occurrence of pesticides in organically
1942 managed agricultural soils - the ghost of a conventional agricultural past? *Environ Sci Technol* 55:2919-
1943 2928. <https://dx.doi.org/10.1021/acs.est.0c06405>

1944 Rios FM, Wilcoxon TE, Zimmerman LM (2017) Effects of imidacloprid on *Rana catesbeiana* immune and nervous
1945 system. *Chemosphere* 188:465-469. <https://doi.org/10.1016/j.chemosphere.2017.08.155>

1946 Rogers MA, Krischik VA, Martin LA (2007) Effect of soil application of imidacloprid on survival of adult green
1947 lacewing, *Chrysoperla carnea* (Neuroptera : Chrysopidae), used for biological control in greenhouse. *Biol*
1948 *Control* 42(2):172-177. <http://dx.doi.org/10.1016/j.biocontrol.2007.05.006>

1949 Rogers KH, McMillin S, Olstad KJ, Poppenga RH (2019) Imidacloprid poisoning of songbirds following a drench
1950 application of trees in a residential neighborhood in California, USA. *Environ Tox Chem* 38(8):1724-1727.
1951 <http://dx.doi.org/10.1002/etc.4473>

1952 Rolke D, Fuchs S, Grunewald B, Gao ZL, Blenau W (2016) Large-scale monitoring of effects of clothianidin-
1953 dressed oilseed rape seeds on pollinating insects in Northern Germany: effects on honey bees (*Apis*
1954 *mellifera*). *Ecotoxicology* 25(9):1648-1665. <http://dx.doi.org/10.1007/s10646-016-1725-8>

1955 Rosas-Ramos N, Banos-Picon L, Tormos J, Asis JD (2020) Natural enemies and pollinators in traditional cherry
1956 orchards: Functionally important taxa respond differently to farming system. *Agric Ecosyst Environ*
1957 295:106920. <http://dx.doi.org/10.1016/j.agee.2020.106920>

1958 Rouchaud J, Gustin F, Wauters A (1994) Soil biodegradation and leaf transfer of insecticide imidacloprid applied
1959 in seed dressing in sugar beet crops. *Bull Environ Contam Toxicol* 53(3):344-350.
1960 <https://doi.org/10.1007/BF00197224>

1961 Roy CL, Coy PL, Chen D, Ponder J, Jankowski M (2019) Multi-scale availability of neonicotinoid-treated seed
1962 for wildlife in an agricultural landscape during spring planting. *Sci Total Environ* 682:271-281.
1963 <https://doi.org/10.1016/j.scitotenv.2019.05.010>

1964 Roy CL, Jankowski MD, Ponder J, Chen D (2020) Sublethal and lethal methods to detect recent imidacloprid
1965 exposure in birds with application to field studies. *Environ Tox Chem* 39(7):1355-1366.
1966 <http://dx.doi.org/10.1002/etc.4721>

1967 Roy CL, Chen D (2023) High population prevalence of neonicotinoids in sharp-tailed grouse and greater prairie-
1968 chickens across an agricultural gradient during spring and fall. *Sci Total Environ* 856:159120.
1969 <https://doi.org/10.1016/j.scitotenv.2022.159120>

1970 Rundlöf M, Andersson GKS, Bommarco R, Fries I, Hederstrom V, Herbertsson L, Jonsson O, Klatt BK, Pedersen
1971 TR, Yourstone J, Smith HG (2015) Seed coating with a neonicotinoid insecticide negatively affects wild
1972 bees. *Nature* 521(7550):77-U162. <http://dx.doi.org/10.1038/nature14420>

1973 Sabin LB, Mora MA (2022) Ecological risk assessment of the effects of neonicotinoid insecticides on northern
1974 bobwhites (*Colinus virginianus*) in the South Texas Plains Ecoregion. *Integr Environ Assess Manage* 18:488-
1975 499. <https://doi.org/10.1002/ieam.4479>

1976 Saito T, Doi M, Katayama H, Kaneko S, Tagami Y, Sugiyama K (2008) Seasonal abundance of hymenopteran
1977 parasitoids of the leafminer *Chromatomyia horticola* (Diptera: Agromyzidae) and the impact of insecticide
1978 applications on parasitoids in garden pea field. *Appl Entomol Zool* 43(4):617-624.
1979 <http://dx.doi.org/10.1303/aez.2008.617>

1980 Samson-Robert O, Labrie G, Chagnon M, Fournier V (2017) Planting of neonicotinoid-coated corn raises honey
1981 bee mortality and sets back colony development. *Peerj* 5:e3670. <http://dx.doi.org/10.7717/peerj.3670>

1982 Samojeden CG, Pavan FA, Rutkoski CF, Folador A, da Fré SP, Müller C, Hartmann PA, Hartman M (2022)
1983 Toxicity and genotoxicity of imidacloprid in the tadpoles of *Leptodactylus luctator* and *Physalaemus cuvieri*
1984 (*Anura: Leptodactylidae*). *Scient Rep* 12:11926. <https://doi.org/10.1038/s41598-022-16039-z>

1985 Sánchez-Bayo F, Goka K (2005) Unexpected effects of zinc pyrethrin and imidacloprid on Japanese medaka fish
1986 (*Oryzias latipes*). *Aquatic Toxicol* 74:285-293. <https://doi.org/10.1016/j.aquatox.2005.06.003>

1987 Sánchez-Bayo F, Goka K, Hayasaka D (2016) Contamination of the aquatic environment with neonicotinoids and
1988 its implication for ecosystems. *Front Environ Sci* 4:71. <http://dx.doi.org/10.3389/fenvs.2016.00071>

1989 Sandrock C, Tanadini LG, Pettis JS, Biesmeijer JC, Potts SG, Neumann P (2014) Sublethal neonicotinoid
1990 insecticide exposure reduces solitary bee reproductive success. *Agric For Entomol* 16:119-128.
1991 <https://doi.org/10.1111/afe.12041>

1992 Sappington JD (2018) Imidacloprid alters ant sociobehavioral traits at environmentally relevant concentrations.
1993 *Ecotoxicology* 27:1179-1187. <https://doi.org/10.1007/s10646-018-1976-7>

1994 Sarnaik SS, Kanekar PP, Raut VM, Taware SP, Chavan KS, Bhadbhade BJ (2006) Effect of application of different
1995 pesticides to soybean on the soil microflora. *J Environ Biol* 27(2):423-426.

1996 Schaafsma AW, Limay-Rios V, Baute TS, Smith JL (2019) Neonicotinoid insecticide residues in subsurface
1997 drainage and open ditch water around maize fields in southwestern Ontario. *Ploce One* 14(4): e021478.
1998 <https://doi.org/10.1371/journal.pone.0214787>

1999 Schepker TJ, Webb EB, Tillitt D, LaGrange T (2020) Neonicotinoid insecticide concentrations in agricultural
2000 wetlands and associations with aquatic invertebrate communities. *Agric Ecosyst Environ* 287:106678.
2001 <http://dx.doi.org/10.1016/j.agee.2019.106678>

2002 Schmidt TS, Miller JL, Mahler BJ, Van Metre PC, Nowell LH, Sandstrom MW, Carlisle DM, Moran PW, Bradley
2003 PM (2022) Ecological consequences of neonicotinoid mixtures in streams. *Science Adv* 8:1-12.
2004 <https://doi.org/10.1126/sciadv.abj8182>

2005 Schott M, Sandmann M, Cresswell JE, Becher MA, Eichner G, Brandt DT, Halitschke R, Krueger S, Morlock G,
2006 During RA, Vilcinskis A, Meixner MD, Buchler R, Brandt A (2021) Honeybee colonies compensate for
2007 pesticide-induced effects on royal jelly composition and brood survival with increased brood production.
2008 *Scient Rep* 11:62. <http://dx.doi.org/10.1038/s41598-020-79660-w>

2009 Sgolastra F, Medrzycki P, Bortolotti L, Renzi MT, Tosi S, Bogo G, Teper D, Porrini C, Molowny-Horas R, Bosch
2010 J (2017) Synergistic mortality between a neonicotinoid insecticide and an ergosterol-biosynthesis-inhibiting
2011 fungicide in three bee species. *Pest Manage Sci* 73(6):1236-1243. <http://dx.doi.org/10.1002/ps.4449>

2012 Shahid N, Becker JM, Krauss M, Brack W, Liess M (2018a) Adaptation of *Gammarus pulex* to agricultural
2013 insecticide contamination in streams. *Sci Total Environ* 621:479-485.
2014 <http://dx.doi.org/10.1016/j.scitotenv.2017.11.220>

2015 Shahid N, Becker JM, Krauss M, Brack W, Liess M (2018b) Pesticide body burden of the crustacean *Gammarus*
2016 *pulex* as a measure of toxic pressure in agricultural streams. *Environ Sci Technol* 52(14):7823-7832.
2017 <http://dx.doi.org/10.1021/acs.est.8b01751>

2018 Shaw PW, Wallis DR (2010) Susceptibility of the European earwig, *Forficula auricularia*, to insecticide residues
2019 on apple leaves. *N Z Plant Prot* 63:55-59.

2020 Shinya S, Sashika M, Minamikawa M, Itoh T, Yohannes YB, Nakayama SMM, Ishizuka M, Nimako C, Ikenaka
2021 Y (2022) Estimation of the effects of neonicotinoid insecticides on wild raccoon, *Procyon lotor*, in Hokkaido,
2022 Japan: Urinary concentrations and hepatic metabolic capability of neonicotinoids. *Environ Toxicol Chem*
2023 41:1865-1874. <https://doi.org/10.1002/etc.5349>

2024 Shukla S, Jhamtani RC, Dahiya MS, Agarwal R (2017) Oxidative injury caused by individual and combined
2025 exposure of neonicotinoid, organophosphate and herbicide in zebrafish. *Toxicol Rep* 4:240-244.
2026 <https://doi.org/10.1016/j.toxrep.2017.05.002>

2027 Sievers M, Hale R, Swearer SE, Parris KM (2018) Contaminant mixtures interact to impair predator-avoidance
2028 behaviours and survival in a larval amphibian. *Ecotox Environ Safe* 161:482-488.
2029 <https://doi.org/10.1016/j.ecoenv.2018.06.028>

2030 Silva V, Mol HGJ, Zomer P, Tienstra M, Ritsema CJ, Geissen V (2019) Pesticide residues in European agricultural
2031 soils – a hidden reality unfolded. *Sci Total Environ* 653:1532-1545.
2032 <https://doi.org/10.1016/j.scitotenv.2018.10.441>.

2033 Simon-Delso N, Amaral-Rogers V, Belzunces LP, Bonmatin JM, Chagnon M, Downs C, Furlan L, Gibbons DW,
2034 Giorio C, Girolami V, Goulson D, Kreutzwiser DP, Krupke CH, Liess M, Long E, McField M, Mineau P,
2035 Mitchell EAD, Morrissey CA, Noome DA, Pisa L, Settele J, Stark JD, Tapparo A, Van Dyck H, Van Praagh
2036 J, Van der Sluijs JP, Whitehorn PR, Wiemers M (2015) Systemic insecticides (neonicotinoids and fipronil):
2037 trends, uses, mode of action and metabolites. *Environ Sci Pollut Res* 22:5-34.
2038 <http://dx.doi.org/10.1007/s11356-014-3470-y>

2039 Sol Dourdin T, Rivière G, Cormier A, Di Poi C, Guyomard K, Rabiller M, Akcha F, Bah Sadialiou T, Le Monier
2040 P, Sussarellu R (2023) Molecular and phenotypic effects of early exposure to an environmentally relevant
2041 pesticide mixture in the Pacific oyster, *Crassostrea gigas*. *Environ Pollut* 326:121472.
2042 <https://doi.org/10.1016/j.envpol.2023.121472>

2043 Stahlschmidt P, Brühl CA (2012) Bats at risk? Bat activity and insecticide residue analysis of food items in an
2044 apple orchard. *Environ Tox Chem* 31(7):1556-1563. <http://dx.doi.org/10.1002/etc.1834>

2045 Stahlschmidt P, Hahn M, Brühl CA (2017) Nocturnal risks-high bat activity in the agricultural landscape indicates
2046 potential pesticide exposure. *Front Environ Sci* 5:62. <http://dx.doi.org/10.3389/fenvs.2017.00062>

2047 Stehle S, Bub S, Schulz R (2018) Compilation and analysis of global surface water concentrations for individual
2048 insecticide compounds. *Sci Total Environ* 639:516-525. <https://doi.org/10.1016/j.scitotenv.2018.05.158>

2049 Strobl V, Albrecht M, Villamar-Bouza L, Tosi S, Neumann P, Straub L (2021a) The neonicotinoid thiamethoxam
2050 impairs male fertility in solitary bees, *Osmia cornuta*. *Environ Pollut* 284:117106.
2051 <http://dx.doi.org/10.1016/j.envpol.2021.117106>

2052 Strobl V, Bruckner S, Radford S, Wolf S, Albrecht M, Villamar-Bouza L, Maitip J, Kolari E, Chantawannakul P,
2053 Glauser G, Williams GR, Neumann P, Straub L (2021b) No impact of neonicotinoids on male solitary bees
2054 *Osmia cornuta* under semi-field conditions. *Physiological Entomology* 46(1):105-109.
2055 <http://dx.doi.org/10.1111/phen.12349>

2056 Stuligross C, Williams NM (2020) Pesticide and resource stressors additively impair wild bee reproduction. *Proc*
2057 *Royal Soc B* 287:20201390. <https://doi.org/10.1098/rspb.2020.1390>

2058 Stuligross C, Williams NM (2021) Past insecticide exposure reduces bee reproduction and population growth rate.
2059 *Proc Natio Acad Sci USA* 118:e2109909118. <https://doi.org/10.1073/pnas.2109909118>

2060 Sumon KA, Ritika AK, Peeters ET, Rashid H, Bosma RH, Rahman MS, Fatema MK, Van den Brink PJ (2018)
2061 Effects of imidacloprid on the ecology of sub-tropical freshwater microcosms. *Environ Pollut* 236:432-441.
2062 <http://dx.doi.org/10.1016/j.envpol.2018.01.102>

2063 Sur R, Stork A (2003). Uptake, translocation and metabolism of imidacloprid in plants. *Bull Insectol* 56:35-40.

2064 Suryanarayanan S (2013) Balancing control and complexity in field studies of neonicotinoids and honey bee
2065 health. *Insects* 4(1):153-167. <https://www.mdpi.com/2075-4450/4/1/153>

2066 Švara V, Krauss M, Michalski SG, Altenburger R, Brack W, Luckenbach T (2021) Chemical pollution levels in a
2067 river explain site-specific sensitivities to micropollutants within a genetically homogeneous population of
2068 freshwater amphipods. *Environ Sci Technol* 55(9):6087-6096. <http://dx.doi.org/10.1021/acs.est.0c07839>

2069 Sweeney MR, Thompson CM, Popescu VD (2021) Sublethal, behavioral, and developmental effects of the
2070 neonicotinoid pesticide imidacloprid on larval wood frogs (*Rana sylvatica*). *Environ Tox Chem* 40(7):1840-
2071 1849. <https://doi.org/10.1002/etc.5047>

2072 Szczepaniec A, Creary SF, Laskowski KL, Nyrop JP, Raupp MJ (2011) Neonicotinoid insecticide imidacloprid
2073 causes outbreaks of spider mites on elm trees in urban landscapes. *Plos One* 6:e20018.
2074 <https://doi.org/10.1371/journal.pone.0020018>

2075 Taliansky-Chamudis A, Gomez-Ramirez P, Leon-Ortega M, Garcia-Fernandez AJ (2017) Validation of a
2076 QuEChERS method for analysis of neonicotinoids in small volumes of blood and assessment of exposure in

2077 Eurasian eagle owl (*Bubo bubo*) nestlings. *Sci Total Environ* 595:93-100.
2078 <http://dx.doi.org/10.1016/j.scitotenv.2017.03.246>

2079 Tapie N, Budzinski H (2018) Quantification de la présence dans les eaux bilan de 2010 à 2016 Rapport REPAR
2080 (Reseau Pesticides du Bassin d’Arcachon), 25.

2081 Tapparo A, Giorio C, Marzaro M, Marton D, Solda L, Girolami V (2011) Rapid analysis of neonicotinoid
2082 insecticides in guttation drops of corn seedlings obtained from coated seeds. *J Environ Monitor* 13(6):1564-
2083 1568. <https://doi.org/10.1039/c1em10085h>

2084 Tennekes H, Zillweger AB (2010) The systemic insecticides: a disaster in the making. Swiss Society of
2085 Toxicology, Annual Meeting, 22 November 2012. ETS Nederland BV Zutphen, 57 p.
2086 [https://www.boerenlandvogels.nl/sites/default/files/Tennekes_Presentation_Annual%20Meeting_Swiss%20](https://www.boerenlandvogels.nl/sites/default/files/Tennekes_Presentation_Annual%20Meeting_Swiss%20Toxicology%20Society_%2022112012.pdf)
2087 [Toxicology%20Society_%2022112012.pdf](https://www.boerenlandvogels.nl/sites/default/files/Tennekes_Presentation_Annual%20Meeting_Swiss%20Toxicology%20Society_%2022112012.pdf). Accessed 20 September 2023

2088 Tetsatsi ACM, Nkeng-Effouet PA, Alumeti DM, Bonsou GRF, Kamanyi A, Watcho P (2019) Colibri (R)
2089 insecticide induces male reproductive toxicity: alleviating effects of *Lansea acida* (Anacardiaceae) in rats.
2090 *Basic Clini Androl* 29:16. <http://dx.doi.org/10.1186/s12610-019-0096-4>

2091 Thiel S, Kohler HR (2016) A sublethal imidacloprid concentration alters foraging and competition behaviour of
2092 ants. *Ecotoxicology* 25:814-823. <https://doi.org/10.1007/s10646-016-1638-6>

2093 Thompson DA, Lehmler HJ, Kolpin DW, Hladik ML, Vargo JD, Schilling KE, LeFevre GH, Peeples TL, Poch
2094 MC, LaDuca LE, Cwiertny DM, Field RW (2020) A critical review on the potential impacts of neonicotinoid
2095 insecticide use: current knowledge of environmental fate, toxicity, and implications for human health.
2096 *Environ Sci: Processes Impacts* 22:1315-1346. <https://doi.org/10.1039/C9EM00586B>

2097 Thompson CM, Sweeney MR, Popescu VD (2022) Carryover effects of pesticide exposure and pond drying on
2098 performance, behavior, and sex ratios in a pool breeding amphibian. *J Zool* 317:229-240.
2099 <https://doi.org/10.1111/jzo.12975>

2100 Tison L, Roessner A, Gerschewski S, Menzel R (2019) The neonicotinoid clothianidin impairs memory processing
2101 in honey bees. *Ecotox Environ Safe* 180:139-145. <http://dx.doi.org/10.1016/j.ecoenv.2019.05.007>

2102 Tomizawa M (2004) Neonicotinoids and derivatives: Effects in mammalian cells and mice. *J Pestic Sci* 29:177-
2103 183. <https://doi.org/10.1584/jpestics.29.177>

2104 Tooker JF, Pearsons KA (2021) Newer characters, same story: neonicotinoid insecticides disrupt food webs
2105 through direct and indirect effects. *Curr Op Insect Sci* 46:50-56. <https://doi.org/10.1016/j.cois.2021.02.013>

2106 Topal A, Alak G, Ozkaraca M, Yeltekin AC, Comaklı S, Acil G, Kokturk M, Atamanalp (2017) Neurotoxic
2107 responses in brain tissues of rainbow trout exposed to imidacloprid pesticide: Assessment of 8-hydroxy-2-
2108 deoxyguanosine activity, oxidative stress and acetylcholinesterase activity. *Chemosphere* 175:186-191.
2109 <https://doi.org/10.1016/j.chemosphere.2017.02.047>

2110 Torres JB, Silva-Torres CSA, Barros R (2003) Relative effects of the insecticide thiamethoxam on the predator
2111 *Podisus nigrispinus* and the tobacco whitefly *Bemisia tabaci* in nectaried and nectariless cotton. *Pest Manage*
2112 *Sci* 59(3):315-323. <http://dx.doi.org/10.1002/ps.640>

2113 Tu C, Wang Y, Duan W, Hertl P, Tradway L, Brandenburg R, Lee D, Snell M, Hu S (2011) Effects of fungicides
2114 and insecticides on feeding behavior and community dynamics of earthworms: Implications for casting
2115 control in turfgrass systems. *Appl Soil Ecol* 47:31-36. <http://dx.doi.org/10.1016/j.apsoil.2010.11.002>

2116 Tyler B (2022) Assessment of duckling abundance as a biological indicator of wetland health in the Prairie Pothole
2117 Region. Master of Science Thesis. <https://harvest.usask.ca/handle/10388/14291> Accessed 20 September
2118 2023

2119 Uhl P, Bucher R, Schäfer RB, Entling MH (2015) Sublethal effects of imidacloprid on interactions in a tritrophic
2120 system of non-target species. *Chemosphere* 132:152-158.
2121 <https://doi.org/10.1016/j.chemosphere.2015.03.027>

2122 Uhl P, Brühl CA (2019) The impact of pesticides on flower-visiting insects: A review with regard to European
2123 risk assessment. *Environ Tox Chem* 38(11):2355-2370. <http://dx.doi.org/10.1002/etc.4572>

2124 van der Sluijs JP, Simon-Delso N, Goulson D, Maxim L, Bonmatin JM, Belzunces LP (2013) Neonicotinoids, bee
2125 disorders and the sustainability of pollinator services. *Curr Op Environ Sustain* 5(3-4):293-305.
2126 <https://doi.org/10.1016/j.cosust.2013.05.007>

2127 van Dijk TC, Van Staalduinen MA, Van der Sluijs JP (2013) Macro-invertebrate decline in surface water polluted
2128 with imidacloprid. *Plos One* 8(5):10. <http://dx.doi.org/10.1371/journal.pone.0062374>

2129 van Meter RJ, Glinski DA, Henderson WM, Garrison AW, Cyterski M, Purucker ST (2015) Pesticide uptake
2130 across the amphibian Dermis through soil and overspray exposures. *Arch Environ Contam Toxicol*
2131 69(4):545-556. <http://dx.doi.org/10.1007/s00244-015-0183-2>

2132 Vijver MG, van den Brink PJ (2014) Macro-invertebrate decline in surface water polluted with imidacloprid: A
2133 rebuttal and some new analyses. *Plos One* 9(2):9. <http://dx.doi.org/10.1371/journal.pone.0089837>

2134 Vinebrooke RD, Cottingham KL, Norberg J, Scheffer M, Dodson SI, Maberly SC, Sommer U (2004) Impacts of
2135 multiple stressors on biodiversity and ecosystem functioning: the role of species co-tolerance. *Oikos*
2136 104(3):451-457.

2137 Vyas NB (1999) Factors influencing estimation of pesticide-related wildlife mortality. *Toxicol Indust Health* 15(1-
2138 2):187-192. <http://dx.doi.org/10.1177/074823379901500116>

2139 Walters KFA (2016) Neonicotinoids, bees and opportunity costs for conservation. *Insect Conserv Div* 9(5):375-
2140 383. <http://dx.doi.org/10.1111/icad.12177>

2141 Wang K, Pang S, Mu XY, Qi SZ, Li DZ, Cui F, Wang CJ (2015) Biological response of earthworm, *Eisenia fetida*,
2142 to five neonicotinoid insecticides. *Chemosphere* 132:120-126.
2143 <http://dx.doi.org/10.1016/j.chemosphere.2015.03.002>

2144 Wang Y, Yang G, Dai D, Xu Z, Cai L, Wang Q, Yu Y (2017) Individual and mixture effects of five agricultural
2145 pesticides on zebrafish (*Danio rerio*) larvae. *Environ Sci Pollut Res* 24(5) 4528-4536.
2146 <https://doi.org/10.1007/s11356-016-8205-9>

2147 Wang P, Zhou LL, Yang F, Liu XM, Wang Y, Lei CL, Si SY (2018a) Lethal and behavioral sublethal side effects
2148 of thiamethoxam on the predator *Harmonia axyridis*. *Entomol Exp Applic* 166(8):703-712.
2149 <http://dx.doi.org/10.1111/eea.12702>

2150 Wang YH, Zhang Y, Xu P, Guo BY, Li W (2018b) Metabolism distribution and effect of thiamethoxam after oral
2151 exposure in Mongolian Racerunner (*Eremias argus*). *J Agric Food Chem* 66(28):7376-7383.
2152 <http://dx.doi.org/10.1021/acs.jafc.8b02102>

2153 Wang YH, Zhang Y, Li W, Yang L, Guo BY (2019) Distribution, metabolism and hepatotoxicity of neonicotinoids
2154 in small farmland lizard and their effects on GH/IGF axis. *Sci Total Environ* 662:834-841.
2155 <http://dx.doi.org/10.1016/j.scitotenv.2019.01.277>

2156 Wang ZK, Tian ZN, Chen L, Zhang WJ, Zhang LY, Li Y, Diao JL, Zhou ZQ (2020) Stereoselective metabolism
2157 and potential adverse effects of chiral fungicide triadimenol on *Eremias argus*. *Environ Sci Pollut Res*
2158 27(8):7823-7834. <http://dx.doi.org/10.1007/s11356-019-07205-4>

2159 Wang JQ, Yin R, Liu YL, Wang BJ, Wang NN, Xiao PF, Xiao TF, Hirai H (2023) Meta-analysis of neonicotinoid
2160 insecticides in global surface waters. *Environ Sci Pollut Res* 30:1039-1047.
2161 <http://dx.doi.org/10.1007/s11356-022-22270-y>

2162 Warne MSJ, Turner RDR, Davis AM, Smith R, Huang A (2022) Temporal variation of imidacloprid concentration
2163 and risk in waterways discharging to the Great Barrier Reef and potential causes. *Sci Total Environ*
2164 823:153556. <https://doi.org/10.1016/j.scitotenv.2022.153556>

2165 Weston DP, Poynton HC, Wellborn GA, Lydy MJ, Blalock BJ, Sepulveda MS, Colbourne JK (2013) Multiple
2166 origins of pyrethroid insecticide resistance across the species complex of a nontarget aquatic crustacean,
2167 *Hyalella azteca*. *Proc Natio Acad Sci USA*, 110(41):16532-16537. <https://doi.org/10.1073/pnas.1302023110>

2168 Wettstein FE, Kasteel R, Garcia Delgado MF, Hanke I, Huntscha S, Balmer ME, Poiger T, Bucheli TD (2016)
2169 Leaching of the neonicotinoids thiamethoxam and imidacloprid from sugar beet seed dressings to subsurface
2170 tile drains. *J Agric Food Chem* 64(33):6407-6415. <http://dx.doi.org/10.1021/acs.jafc.6b02619>

2171 Whitehorn PR, O'Connor S, Wackers FL, Goulson D (2012) Neonicotinoid pesticide reduces bumble bee colony
2172 growth and queen production. *Science* 336(6079):351-352. <http://dx.doi.org/10.1126/science.1215025>

2173 Wiest L, Bulete A, Giroud B, Fratta C, Amic S, Lambert O, Pouliquen H, Arnaudguilhem C (2011) Multi-residue
2174 analysis of 80 environmental contaminants in honeys, honeybees and pollens by one extraction procedure
2175 followed by liquid and gas chromatography coupled with mass spectrometric detection. *J Chrom A*
2176 1218(34):5743-5756. <https://doi.org/10.1016/j.chroma.2011.06.079>

2177 Wilcox AAE, Newman AEM, Raine NE, Mitchell GW, Norris DR (2021) Effects of early-life exposure to
2178 sublethal levels of a common neonicotinoid insecticide on the orientation and migration of monarch
2179 butterflies (*Danaus plexippus*). *J Exp Biol* 224:jeb230870. <http://dx.doi.org/10.1242/jeb.230870>

2180 Willemsen RE, Hailey A (2001) Effects of spraying the herbicides 2,4-D and 2,4,5-T on a population of the tortoise
2181 *Testudo hermanni* in southern Greece. *Environ Pollut* 113(1):71-78. [http://dx.doi.org/10.1016/s0269-](http://dx.doi.org/10.1016/s0269-7491(00)00160-3)
2182 [7491\(00\)00160-3](http://dx.doi.org/10.1016/s0269-7491(00)00160-3)

2183 Williams N, Sweetman J (2019) Effects of neonicotinoids on the emergence and composition of chironomids in
2184 the Prairie Pothole Region. *Environ Sci Pollut Res* 26:3862-3868. [http://dx.doi.org/10.1007/s11356-018-](http://dx.doi.org/10.1007/s11356-018-3683-6)
2185 [3683-6](http://dx.doi.org/10.1007/s11356-018-3683-6)

2186 Wood TJ, Goulson D (2017) The environmental risks of neonicotinoid pesticides: a review of the evidence post
2187 2013. *Environ Sci Pollut Res* 24(21):17285-17325. <http://dx.doi.org/10.1007/s11356-017-9240-x>

2188 Woodcock BA, Isaac NJB, Bullock JM, Roy DB, Garthwaite DG, Crowe A, Pywell RF (2016) Impacts of
2189 neonicotinoid use on long-term population changes in wild bees in England. *Nature Com* 7:12459.
2190 <https://doi.org/10.1038/ncomms12459>

2191 Woodcock BA, Bullock JM, Shore RF, Heard MS, Pereira MG, Redhead J, Ridding L, Dean H, Sleep D, Henrys
2192 P, Peyton J, Hulmes S, Hulmes L, Sarospataki M, Saure C, Edwards M, Genersch E, Knabe S, Pywell RF
2193 (2017) Country-specific effects of neonicotinoid pesticides on honey bees and wild bees. *Science*
2194 356(6345):1393-1395. <http://dx.doi.org/10.1126/science.aaa1190>

2195 Wu S, Li X, Liu X, Yang G, An X, Wang Q, Wang Y (2018) Joint toxic effects of triazophos and imidacloprid on
2196 zebrafish (*Danio rerio*). *Environ Pollut* 235:470-481. <https://doi.org/10.1016/j.envpol.2017.12.120>

2197 Wu CH, Lin CL, Wang SE, Lu CW (2020) Effects of imidacloprid, a neonicotinoid insecticide, on the echolocation
2198 system of insectivorous bats. *Pestic Biochem Physiol* 163:94-101.
2199 <https://doi.org/10.1016/j.pestbp.2019.10.010>

2200 Wu-Smart J, Spivak M (2018) Effects of neonicotinoid imidacloprid exposure on bumble bee (*Hymenoptera*:
2201 *Apidae*) queen survival and nest initiation. *Environ Entomol* 47(1):55-62.
2202 <http://dx.doi.org/10.1093/ee/nvx175>

2203 Xu W, Zhang L, Hou J, Du X, Chen L (2023) Absorption and distribution of imidacloprid and its metabolites in
2204 Goldfish (*Carassius auratus* Linnaeus). *Toxics* 11:619. <https://doi.org/10.3390/toxics11070619>

2205 Yamamuro M, Komuro T, Kamiya H, Kato T, Hasegawa H, Kameda Y (2019) Neonicotinoids disrupt aquatic
2206 food webs and decrease fishery yields. *Science* 366(6465):620-623. <https://doi.org/10.1126/science.aax3442>

2207 Yang L, Shen Q, Zeng T, Li J, Li W, Wang Y (2020) Enrichment of imidacloprid and its metabolites in lizards
2208 and its toxic effects on gonads. *Environ Pollut* 258:113748. <https://doi.org/10.1016/j.envpol.2019.113748>

2209 Yu B, Chen ZY, Lu XX, Huang YT, Zhou Y, Zhang Q, Wang D, Li JY (2020) Effects on soil microbial community
2210 after exposure to neonicotinoid insecticides thiamethoxam and dinotefuran. *Sci Total Environ* 725:138328.
2211 <http://dx.doi.org/10.1016/j.scitotenv.2020.138328>

2212 Zhang M, Liang YC, Son A, Yu B, Zeng XB, Chen MS, Yin HQ, Zhang XX, Sun BL, Fan FL (2017) Loss of soil
2213 microbial diversity may increase insecticide uptake by crop. *Agric Ecosyst Environ* 240:84-91.
2214 <http://dx.doi.org/10.1016/j.agee.2017.02.010>

2215 Zhang C, Wang X, Kaur P, Gan J (2023) A critical review on the accumulation of neonicotinoid insecticides in
2216 pollen and nectar: Influencing factors and implications for pollinator exposure. *Sci Total Environ*
2217 899:165670. <https://doi.org/10.1016/j.scitotenv.2023.165670>

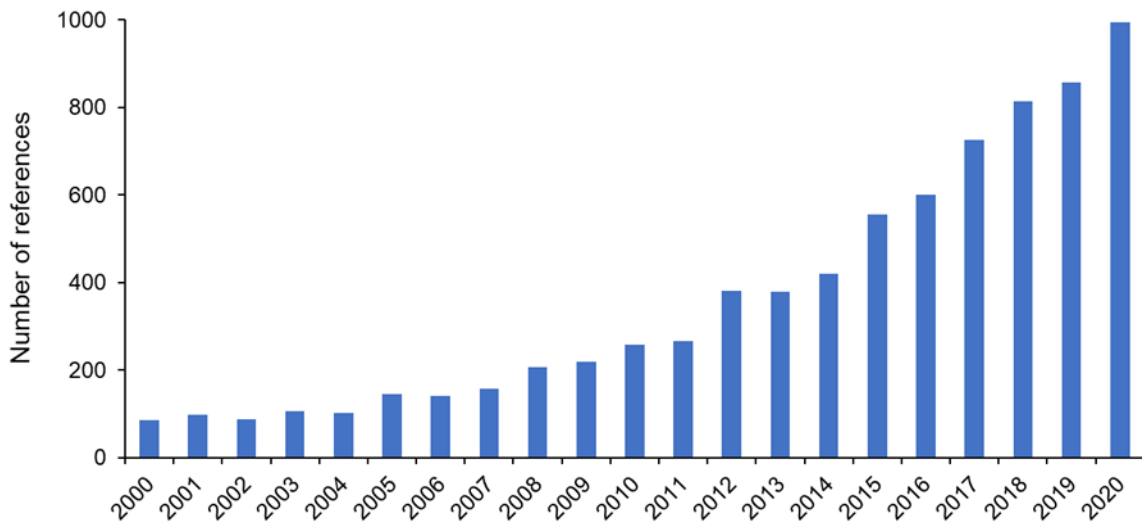
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2221 **Figures**

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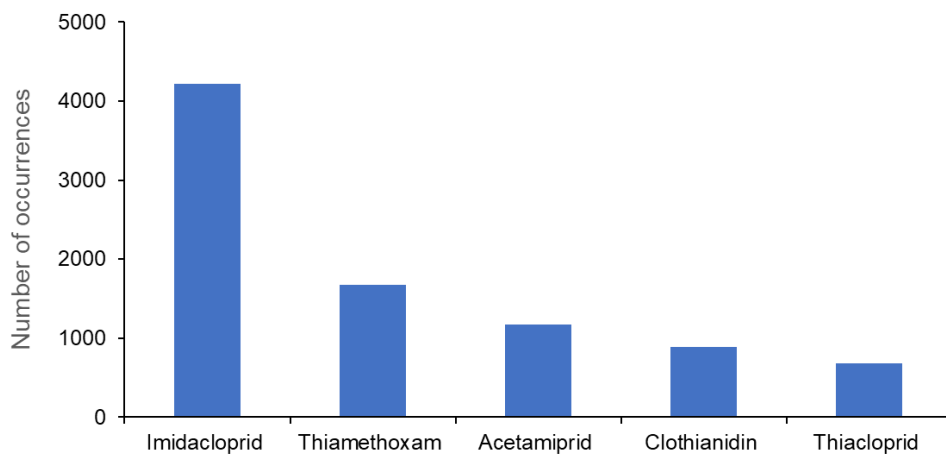


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2224 **Fig. 1** Time course of references focused on the impacts of neonicotinoids on biodiversity.

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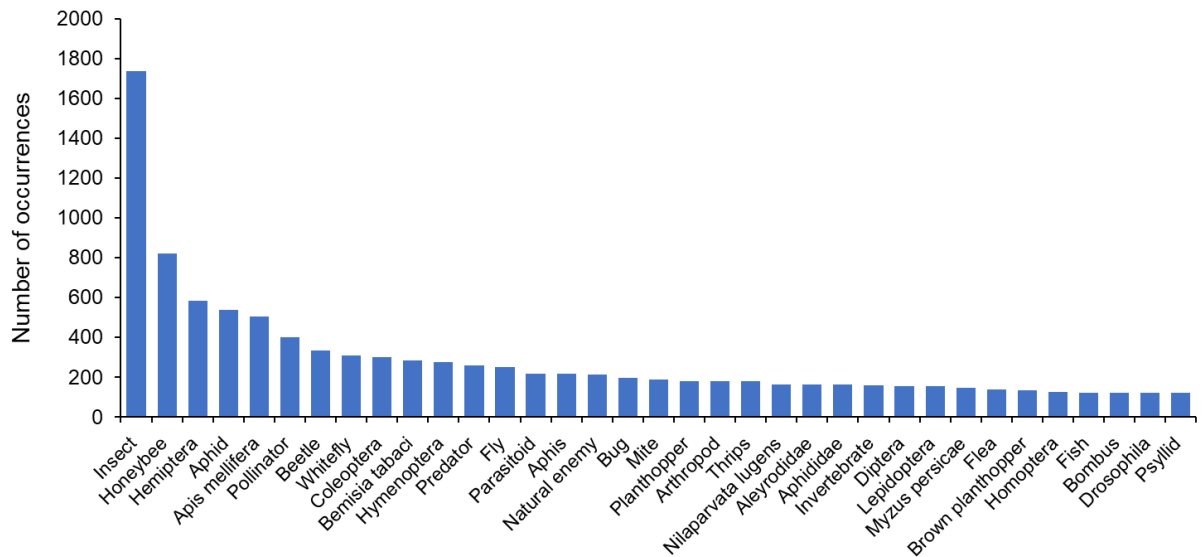
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2229 **Fig. 2** Occurrences of imidacloprid, thiamethoxam, acetamiprid, clothianidin and thiacloprid in title and abstract

2230 of the references constituting the bibliographic corpus on the impacts of neonicotinoids on biodiversity, from 2000

2231 to 2020.

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2234 **Fig. 3** Occurrences of the first 35 organisms studied in the bibliographic corpus on the impacts of neonicotinoids
 2235 on biodiversity, from 2000 to 2020. Occurrences are counted from titles and abstracts. When occurring, alternative
 2236 spellings were gathered into one category, for example “honeybee”, “honey bee”, “honeybees” and “honey bees”.

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2252 **Table**

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2254 **Table 1** Maximum concentration levels of acetamiprid, clothianidin, imidacloprid, thiacloprid and thiamethoxam in soil, air and water observed in France,

2255 Europe and in the world. nd: not determined, *particulate phase.

Neonicotinoid	Geographic zone	Soil		Air		Water	
		Concentration (µg/kg)	Reference	Concentration (ng/m ³)	Reference	Concentration (µg/L)	Reference
Acetamiprid	France	0.48	Froger et al. (2023)	0.26	Phytatmo database (2023)	nd	nd
	Europe	nd	nd	0.031 (Spain)	Coscollà and Yusà (2016)	4.00 (Spain, freshwater)	de Araújo et al. (2022)
	World	nd	nd	0.036* (Canada)	Raina-Fulton (2015)	2.86 (Turkey, freshwater)	de Araújo et al. (2022)
Clothianidin	France	2.7	Froger et al. (2023)	nd	nd	nd	nd
	Europe	57 (Switzerland)	Riedo et al. (2021)	nd	nd	nd	nd
	World	nd	nd	0.09* (Canada)	Raina-Fulton (2015)	3.50 (USA, drained wetlands)	Evelsizer and Skopec (2018)
Imidacloprid	France	160	Pelosi et al. (2021)	2.3	Phytatmo database (2023)	0.132 (USA, freshwater)	de Araújo et al. (2022)
	Europe	138 (Switzerland)	Chiaia-Hernandez et al. (2017)	0.014 (Spain)	Coscollà and Yusà (2016)	2.22 (peri-urban ponds)	Nélieu et al. (2021)
	World	nd	nd	0.36* (Canada)	Raina-Fulton (2015)	0.905 (agricultural/urban rivers)	Criquet et al. (2017)
Thiacloprid	France	1.4	Pelosi et al. (2021)	0.47	Phytatmo database (2023)	0.14 (marine waters)	Auby et al. (2011)
	Europe	14 (Switzerland)	Riedo et al. (2021)	nd	nd	0.342 (Spain, freshwater)	de Araújo et al. 2022
	World	nd	nd	nd	nd	9.14 (USA, freshwater)	Wang et al. (2023)
Thiamethoxam	France	2.0	Pelosi et al. (2021)	0.06	Phytatmo database (2023)	nd	nd
	Europe	24 (Switzerland)	Riedo et al. (2021)	nd	nd	0.159 (Portugal, freshwater)	de Araújo et al. (2022)
	World	nd	nd	nd	nd	1.37 (Australia, lagoon)	Wang et al. (2023)
						0.0039 (bay)	Tapie and Budzinski (2018)
						0.215 (Portugal, freshwater)	de Araújo et al. (2022)
						6.90 (USA, drained wetlands)	Evelsizer and Skopec (2018)
						3.82 (Canada, freshwater)	Wang et al. (2023)

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