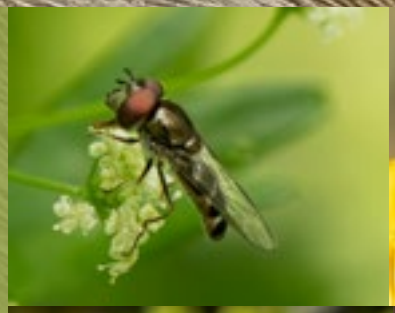
AMERICAN BIRD CONSERVANCY

Neonicotinoid insecticides:

Failing to come to grips with a predictable environmental disaster



Neonicotinoid insecticides: Failing to come to grips with a predictable environmental disaster

American Bird Conservancy, June 2023

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Acknowledgments

The authors have benefitted from discussions with a large number of scientists too numerous to name here. Specific thanks are extended to the Center for Food Safety, Natural Resources Defense Council, and Canadian Wildlife Federation among others, whose support over the last decade has allowed the senior author to remain abreast of the evolving science. We wish to thank Laura Addy-Orduna for sending both published and in press articles; Maria Belen Poliserpi also was kind enough to share both publications as well as hard to get references. We are indebted also to Dr. Christy Morrissey for sharing her 2022 'Notice of objection' to PMRA's final decision on imidacloprid.

About American Bird Conservancy

American Bird Conservancy is a non-profit organization dedicated to conserving wild birds and their habitats throughout the Americas. With an emphasis on achieving results and working in partnership, we take on the greatest problems facing birds today, innovating and building on rapid advancements in science to halt extinctions, protect habitats, eliminate threats, and build capacity for bird conservation. Find us at abcbirds.org or on social media platforms including Facebook, Instagram, Twitter, and LinkedIn.

Executive Summary

Neonicotinoid insecticides ("neonics") were introduced in the 1990s in response to widespread pest resistance as well as health objections to older pesticides. They quickly gained dominance in global pesticide markets and remain the most widely-used insecticides in parts of the world where they have not been banned. In North America, it is difficult to find crop plants that do not contain one or several of the neonicotinoid insecticides.

In 2013, American Bird Conservancy produced a ground-breaking report "The Impact of the Nation's Most Widely Used Insecticides on Birds" (Mineau and Palmer 2013) outlining the risks that this new class of mobile and persistent insecticides placed on both terrestrial and aquatic ecosystems and the likely ripple effects on the wildlife species that depend on these ecosystems, notably birds. At the time, most of the controversy surrounding this class of products concerned honey bee mortality incidents and the potential for neonicotinoids to interfere with crop pollination.

The conclusions of the 2013 report were clear: Some neonicotinoid insecticides, including the dominant product at the time, imidacloprid, are potentially lethal to birds and have the potential to seriously disrupt reproduction and normal behaviour and physiology. Because of their use as seed treatments, exposure routes are plentiful and lead to substantial exposure. Neonicotinoids were also predicted to impact aquatic systems on which birds heavily depend for food. Observed contamination levels in surface and groundwater in the US and around the world were strikingly high, and already beyond the threshold found to kill many aquatic invertebrates. United States Environmental Protection Agency (EPA) risk assessments using scientifically unsound and, outdated methodology, had greatly underestimated this risk.

Given the above, recommendations from American Bird Conservancy and its partners in the National Pesticide Reform Coalition were unequivocal. They urged the EPA to:

- *"Suspend all applications of neonicotinoids pending independent review of these products' effects on birds, terrestrial and aquatic invertebrates, and other wildlife.*
- *Expand its re-registration review of neonicotinoids beyond bees to include birds, aquatic invertebrates, and other wildlife.*
- *Ban the use of neonicotinoids as seed treatments.*
- *Require that registrants of acutely toxic pesticides develop the tools necessary to diagnose poisoned birds and other wildlife."*

This was 2013. Ten years later, it is appropriate to ask what has happened in the interim. Unfortunately, neonicotinoids are still being abundantly used in most crops in North America. None of the new science has dispelled earlier fears of major impacts, and regulators have yet to meet their mandate to accurately weight the costs and benefits of this class of insecticides.

In this report, we look at the science that has developed over the last decade as well as the regulatory response by the EPA and other regulators to address our increasing knowledge base. The key messages to come from this report are taken from the report sections and repeated here. For easy reference and help the reader, the report section that elaborates the point is provided which each key message.

1.1. New toxicity studies carried out over the last decade have reinforced the high acute toxicity to birds of the first-generation neonicotinoids imidacloprid and acetamiprid. A median lethal dose for a sensitive species at the 5 percent tail of a sensitivity distribution (Hazardous Dose $_5$ or HD_5) is calculated to be 8 and 10 mg/kg body weight for acetamiprid and imidacloprid respectively. Scaling the acute toxicity to bodyweight to account for the higher sensitivity of small bird species to acute pesticide ingestion yields an estimated HD_5 value as low as 1 mg/kg for imidacloprid. This potentially brings those neonics in the same range as older neurotoxic insecticides viz. organophosphorus and carbamate products. In the decade following our original report, EPA regulators have not amended their faulty assessment of neonic bird toxicity even though they now use distributional methods to assess the toxicity of neonics to other organisms such as aquatic invertebrates. This largely invalidates any EPA effort to carry out a scientifically tenable risk assessment in birds. In this section we also document doses of incipient mortality or severe physical debilitation. We continue to believe that preventing large numbers of birds from being debilitated on farm fields is the correct biological and ethical standard against which we should hold pesticides to account. A number of independent studies with imidacloprid have shown that our derived 'debilitation threshold' may still be under-protective with various forms of impairment (e.g. escape responses, disrupted energy budgets, migratory ability etc.) being seen at this dose level or lower.

1.2. Neonics, especially the high-volume compounds imidacloprid, clothianidin and thiamethoxam, do not show the extent of differentiation among compounds for chronic and reproductive toxicity in birds as they do for acute toxicity. The original industry studies extensively reviewed in our 2013 report have not changed; new independent studies, however, give credence to our derivation of a reproductive benchmark in the form of an 'MATC' (the geometric mean of no-effect and low effect levels expressed here as mg/kg bodyweight/day) extrapolated to a reasonably sensitive bird (one at the 5 percent tail of an acute toxicity sensitivity distribution). In particular, several new studies with imidacloprid have shown effect levels at very similar or lower doses. Based on recent studies, we have increasing concerns over reproductive and sub-lethal effects resulting from low exposures in farm fields. In particular, impacts on sperm quality have been seen at dose levels a fraction of our calculated MATC; viz. 0.05 percent of the reproductive threshold for acetamiprid; 1.1 percent of the reproductive threshold for clothianidin. Given that exposure is often season-long, this raises the specter of significant effects on a large number of bird species.

1.3. Neonics are not repellant chemicals. Birds may avoid lethal poisoning if 1) their rate of intake of contaminated food items is low enough that they will be made too sick to eat before ingesting a lethal dose; 2) they can recognize what food item has sickened them so that it can be avoided; 3) there are no other 'hidden' sources of exposure beyond their direct control such as drinking water, dermal contact of inhalation of fine droplets or vapors. Learned avoidance in laboratory settings has been found to be highly variable and dependant on test conditions. This is not a new finding. Similar results were seen with older seed products such as organophosphorus insecticides. Learned avoidance is not foolproof and there is often mortality even when the population response shows avoidance. In order for learned avoidance to work, birds have to be made sick and anorexic, often for several hours or up to a day post exposure. Given the scale of use of neonics, is it acceptable to have millions of birds being sickened, made anorexic, paralyzed or exhibiting convulsions following exposure? How many of those birds will appear to leave the fields unscathed to seek cover and die in the following days. This scenario, which was all too common with cholinesterase-inhibiting compounds of medium toxicity is a probable one here, at least with the first generation neonics.

Treated seeds remain an easy route of exposure for birds. Indeed, many farmland species have become dependent on seeds. Regardless of the seeding technology, there are always small spills of

concentrated surface seeds available to birds. This is standard farming practice and it is cynical to think that this can be avoided. The answer is not to blame or criminalize the farmer but rather to ensure that seed treatments are safe for birds and other wildlife.

Observational studies (at least those not carried out by the manufacturers) show that many species are attracted to and will feed on treated seeds, especially when available at spills. Numerous kills have been reported with imidacloprid in Europe, North America and elsewhere.

Residue analyses have confirmed that exposure of birds to neonics are commonplace. Although they are rapidly cleared from tissues in vertebrates, surveys of a broad range of species show that a large proportion of most species examined show evidence of exposure. Gamebirds and seed-eaters frequenting farmland are clearly massively exposed; however, residues are being seen in raptors, hummingbirds and even seabirds. Neonics may not bioaccumulate in organisms and biomagnify in food webs as did DDT and other organochlorine pesticides of old, but they appear to be as widely distributed in the broader terrestrial and aquatic environments.

1.4. Any fairly conducted risk assessment will show that the kills that have been recorded with imidacloprid were totally expected. Calculations show that only a few seeds or a fraction of daily food intake, whether through treated seed or invertebrates in a sprayed field, can prove lethal. Another reason we should not be surprised about kills being likely is that the toxicity of first generation neonics is similar to that of several cholinesterase-inhibiting insecticides that have a track record of kills at equivalent or lower toxic potencies.

Worse yet, birds are expected to experience severe sub-lethal and reproductive effects at levels that are likely, either through the ingestion of treated seeds over a short period of time or through ingestion of contaminated foods such as invertebrates. Documenting the risk of reproductive effects through ingestion of contaminated insects requires that residues be followed over time in the food source. This is beyond our scope here but, it is clear that some of the effect levels obtained from the literature, especially those having to do with sperm quality will easily be breached for a long period of time by a foraging insectivore. This is the type of risk assessment we should be demanding from regulators. Where birds are obtaining residues from insects or vegetation, it becomes impossible for them to associate any post-ingestion illness with a specific food item … such as a colored seed. Insectivorous bird species are not going to stop eating insects even if the latter cause them discomfort. The very large scale of neonic use in our farm fields makes it unlikely that the birds in question can 'move next door' and avoid continued contamination by one neonic or another.

2.1. Our understanding of the importance of insects (and other invertebrates) to birds has evolved in the last decade as more attention is placed on the quality of insect food as well as the quantity. Emergent insects have been shown to be very important form a quality point of view.

2.2. It has become increasingly clear that we are seeing a generalized decline in insects. This will have clear repercussions for consumer species such as birds. In agricultural environments, there are now a number of studies linking neonic use specifically to declines or extirpation of terrestrial insect species such as wild bee species and butterflies.

2.3. In our 2013 report, we commented on the ease with which neonics can move with water – whether surface runoff or contamination of aquifers. What has also become amply clear is the extent to which terrestrial environments a long distance from treated fields can also be contaminated.

The idea that pollinators can be protected by restricting the use of neonics to crops not visited by pollinators (a common strategy employed by regulators) has been shown to be highly inadequate. Field borders and other 'protected' areas that are essential for pollinators (and multitudes of beneficial insects) are typically heavily impacted when crops are treated nearby. There is currently 'no place to hide' from neonics in our farm fields.

2.4. North American regulators, namely EPA, the Canadian Pesticide Management Regulatory Agency (PMRA) and California Department of Pesticide Regulation (DPR) have attempted to assess the risk of the three high-use neonics to pollinators. Despite considerable time and resources devoted to this assessment, the end result is scientifically highly questionable. Although reviewed more extensively elsewhere (Mineau 2020) the summary of that assessment is repeated below. One key finding is that the risk of seed treatments – typically a large proportion of total use – for pollinators at large has been completely mischaracterized by the regulators through a series of fundamental flaws in the official assessment.

2.5. The critical situation in aquatic systems remains essentially as we described it in our 2013 report. Regulators (EPA and PMRA) continue to tinker with aquatic quality benchmarks (presumably in response to industry comments) while ignoring much of the information now clearly linking neonic contamination with widespread disruptions of aquatic ecosystems. Earlier improvements in EPA methodology now appear to have been lost as the Agency appears to have gone back on its assessment of imidacloprid in the context of Endangered Species Act (ESA)-listed species.

Neonics continue to be detected at concentrations that are expected to cause severe impacts in receiving waters. If a benchmark concentration is needed, then the older benchmark of 0.01 µg/L established by EPA for imidacloprid in 2016 should still be the one against which summed concentrations of neonics should be compared. There is still too little evidence that the other neonics are much safer and too many cases where multiple compounds are detected in the same samples. Increasing reports of potential synergisms between different neonics are clearly problematic as even simple additivity of effects is already proving to be a problem in the real world.

Regulators continue to consider one neonic at the time when mixtures are becoming the norm; they continue to ignore the time-cumulative effect of toxicity despite the fact that aquatic contamination is often season-long. In short, regulators should listen to the scientific community and their consensus that neonics need to be severely restricted and water contamination reduced.

3.0. The unwillingness of regulators to fairly assess neonics and reduce their ecosystem-wide impacts continues to baffle. Credible region- or nation-wide analyses now link declines in bird populations to the use of neonic insecticides. An unprecedented 2023 Europe-wide analysis shows that agricultural intensification, as measured by the use of pesticides and fertilizers, is the main driver of bird population declines. The direct link between the inputs associated with intensive agriculture and bird declines is clear. The role of neonics in the intensification of insect control is just as clear.

We are now at the point where the onus of proof should switch from having to demonstrate the link between neonicotinoid use and bird populations losses, to showing why the continued profligate use of neonics is essential to human welfare in light of such environmental impacts. Regulators in North America claim to objectively review the available science and use weight of evidence approaches. Unfortunately, we believe they have failed in the execution of their mandate and in preventing the ongoing environmental tragedy that neonics represent.

4.1. In the European Union and a few Canadian provinces, neonicotinoids are much more heavily scrutinized and regulated. The EU has taken major steps in banning outdoor uses of neonics and, more recently the emergency uses of neonics were overturned in a court, fully banning their use. Canadian provinces have taken a more surgical approach recently by requiring specific prescriptions for neonic use on corn seed, leading to a massive decline in their use.

4.2. The most common use of neonicotinoids in the United States is as a seed coating; yet due to a loophole in federal regulatory law, they are regulated (or, not regulated) as exempted pesticides. The loophole, entitled the Treated Article Exemption, allows voluminous use of neonics without proper oversight from the Environmental Protection Agency. The lack of adequate coated seed oversight has led to an underrepresentation of impacts on wildlife and communities from neonicotinoids. Highly regarded and relied upon sources of information, such as the North American Water Quality Assessment (NAWQA) pesticide use maps, make specific note that seed coatings are too difficult to reliably source information on and, therefore, are not included in national pesticide-use estimates. Despite the dearth of information, there is not a perceived need by federal regulators to reclassify them.

4.3. When EPA does not properly assess the toxicity of a pesticide to exposed species and does not quantitatively account for most of that pesticide's use, how can it possibly arrive at the correct conclusions? The threatened and endangered species (TES) assessment carried out by the EPA represents a monumental effort. However, the assessments of neonic effects on TES are highly under representative of true impacts. It is unfortunate indeed that on both the terrestrial and aquatic side, fundamental errors in EPA's assessment methodology renders the assessment unreliable and under-protective of TES.

4.4. In the absence of adequate federal regulation, some state governments and pesticide oversight bodies have taken steps to limit the use of neonicotinoids. As of May 2023, ten states have enacted legislation making neonicotinoids restricted-use pesticides, meaning only certified pesticide applicators have access to them. However, many of these state laws and regulations include exemptions for agricultural and other intensive uses of pesticides. In certain cases, such as with Nebraska, specific disposal requirements for neonic-coated seeds had to be legislatively enacted after an ecological disaster.

4.5. Neonicotinoids are used at a scale not seen since the introduction of DDT, yet the regulation of these chemicals is severely lacking. Their most voluminous use, as a seed coating, goes mostly unregulated by federal and state entities in any meaningful way. The United States, as is often the case with pesticide regulation, is distantly trailing the European Union (EU) and a few Canadian provinces (Ontario and Québec) in responsible neonicotinoid regulation and mitigation. The main uses of neonicotinoid insecticides go against fundamental principles of integrated pest management. Alternatives to these chemicals do exist.

While the EPA is certainly taking great strides in addressing the presented issues, their efforts are not moving at a pace sufficient to fully mitigate the effects of neonicotinoids on wildlife and ecosystems. Furthermore, until pesticide-coated seeds are reclassified as pesticides and removed from FIFRA exemption, there can be no hope of adequately addressing the risks posed by all uses of neonicotinoids.

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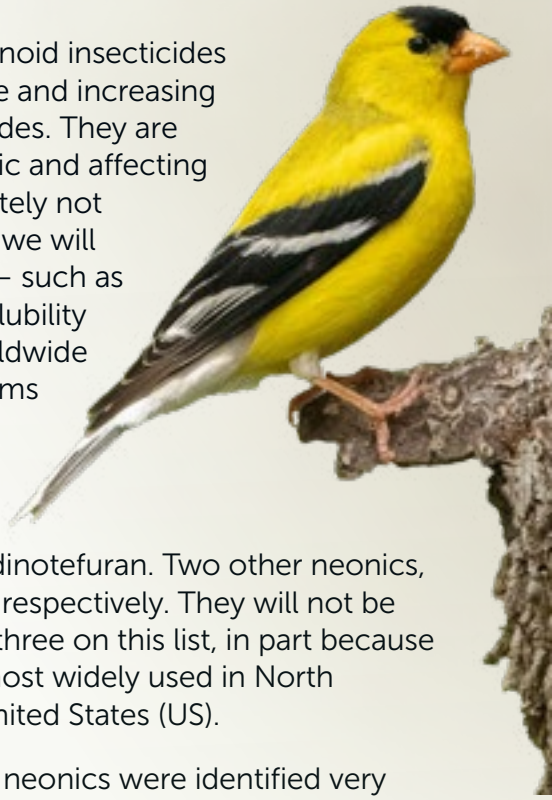
Introduction



As described in our 2013 report (Mineau and Palmer 2013), neonicotinoid insecticides were introduced in the 1990s to counter widespread pest resistance and increasing health and environmental objections to the organophosphorus insecticides. They are generally of lower acute toxicity to vertebrates although being neurotoxic and affecting the same biochemical pathways (the cholinergic system), they are definitely not devoid of effects on exposed vertebrate species (Gibbons et al. 2014) as we will explore in more detail in this report. Of course, their longer persistence – such as half lives in soil extending to several years – and their very high water solubility has meant that they have become ubiquitous in freshwater systems worldwide (Morrissey et al. 2015). This extensive contamination of aquatic ecosystems as much as their impact on pollinator species is what has created the greatest concerns and led to their banning in the (EU).

The neonicotinoid insecticides used for crop protection include imidacloprid, clothianidin, thiamethoxam, thiacloprid, acetamiprid, and dinotefuran. Two other neonics, nitenpyram and nithiazine are used for flea control in pets and in fly bait respectively. They will not be considered any further here. Most of the attention has been on the first three on this list, in part because of their toxicity and persistence but also because these have been the most widely used in North America. There are currently no registered thiacloprid products in the United States (US).

It is important to mention once again that the problems associated with neonics were identified very early on. As early as 1994, EPA scientists had warned that both acute and chronic aquatic risk triggers had been exceeded for both non-endangered and endangered species exposed to imidacloprid. This first-to-be-registered neonicotinoid was correctly assessed to be highly persistent and mobile (USEPA 1994a, b). In Canada, there was a discussion amongst evaluators as to whether imidacloprid could be rejected based on the 2001 Stockholm Convention on Persistent Organic Pollutants; however, it does not bioaccumulate or biomagnify in a food chain. It does, however, contaminate groundwater quite extensively. In 2007, USEPA scientists also extended concerns to vertebrate wildlife citing potential risks from low chronic exposures (USEPA 2007).

In 2003, clothianidin used on treated seed was predicted to give rise to a chronic risk to birds and mammals, given that consumption of 1-2 seeds only could push them to an exposure level at which reproductive effects were expected (USEPA 2003a, b). USEPA scientists further described clothianidin as persistent and mobile, with '*potential to leach to ground water as well as runoff to surface waters*' (USEPA 2003a).

And then as early as 2008, thiamethoxam was said to have the potential to cause '*direct adverse effects on freshwater invertebrates, birds and mammals*' (USEPA 2008). USEPA also predicted '*structural and functional changes of both the aquatic and terrestrial ecosystems*', a rather extreme but fundamentally correct prediction.

As this report will attest, government reviewers initially had it right. It is therefore pertinent to ask why regulatory agencies in both North America and Europe did not listen to their own scientists but rather allowed the pesticide industry to register neonicotinoids on virtually every agricultural commodity known to humankind and make neonicotinoids the most important class of insecticide worldwide (Jeschke et al. 2011). Even without considering the heavy impacts of neonicotinoids on pollinator species (the main focus of the public uproar over their registration) how could regulatory systems fail to this degree at protecting the environment and 'Trust Resources' such as migratory bird species? Why is there a need,

once again, to review the evidence against several neonicotinoid insecticides and denounce the lack of substantive regulatory action?

In the US, the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) mandates the EPA to weigh the risks and benefits of pest control products. Is it therefore that the benefits of their use completely overwhelmed environmental concerns? The short answer is no! Whereas the systemic activity of neonicotinoids might make them very attractive in the case of some specific hard-to-control pests, the benefits of the largest uses of neonicotinoids are often slight and, at times, nonexistent. A full agronomic review is beyond the scope of this report but see USEPA (2014), Douglas and Tooker (2015), Douglas et al. (2015), and Krupke et al. (2017), Pennsylvania State University Extension (2023) for soybean; Alford and Krupke (2017), North et al. (2017), and Li et al. (2022) for corn; Macfadyen et al. (2014) for cereal; Budge et al. (2015) and Hokkanen et al. (2017) for oilseed crops; Clavet et al. (2014) for turf. Other reviews of the literature such as Center for Food Safety (2014, 2016), Veres et al. (2020), Rowen et al. (2022) arrive at a similar conclusion.

In this report, we intend to revisit some of the points we made in our 2013 assessment, look at the scientific research that has accumulated in the intervening decade and ask whether we are still confident that the advice we gave the USEPA 10 years ago, namely to ban the use of neonicotinoids as seed treatments and to suspend many other uses was appropriate. Whereas the answer to 'what is a neonicotinoid' was easy to answer in 2013, it is becoming increasingly confused. According to a recent review (Thompson et al. 2020), more than 600 neonicotinoid compounds have been synthesized in China alone. Some are registered in that country but not in North America or Europe – yet. Also, a number of 'neonicotinoid-like' compounds have also made an appearance; some of which have a very similar mode of action; e.g. flupyradifurone and sulfoxaflor. Six neonics have been registered and are broadly used in agriculture in the western world; these are the focus of this report. They consist of the 'first generation' compounds imidacloprid, acetamiprid and thiacloprid; second generation compounds clothianidin and thiamethoxam and the latest (so-called third generation) dinotefuran.

In our 2013 report, we reviewed the meteoric rise in these compounds, especially the massive amounts used prophylactically as seed treatments. With the exception of Europe where many uses were banned for many of the environmental reasons we will review in this report, use is still increasing according to recent reviews (e.g. Thompson et al. 2020). These authors estimate from a number of sources that, in the U.S., 100 percent of corn, 95 percent of cotton[1], and 50 percent of soybeans have at least one neonic being used as a seed treatment.[2] Based on our own literature research (Mineau 2020), seed treatments in cereal crops already represented 20-27 percent of the crop a decade ago; similarly dated estimates include 46 percent for sugar beets, 44 percent for sorghum, 45-75 percent for rice, and at least 15 percent for vegetable crops. Using California as an example, the use of seed treatments was possible on 76 percent of the total planted acreage in that state. Ironically, EPA refuses to consider treated seeds to be pesticides; a misguided policy that means much data-gathering possibility is stifled.

Uses of neonics other than seed treatment uses are massive also. In 2016, California reported the use of 474,000 lbs (215,000 kg) of active ingredient applied as a liquid or as a granular formulation or roughly 41g for every planted acre in that state (Mineau 2020). This does not include any of the many domestic, structural or veterinary uses, or any of the seed treatment uses which, if fully implemented could more than double that amount. Knowing what we know about the impacts this is creating, how can we allow this situation to continue?

[1] I estimated 100% in an earlier report (Mineau 2020)

[2] They may also receive other neonic treatments during the growing season

Discussion

1. Direct effects of neonics on birds

1.1. Acute toxicity of neonics to birds

1.1. New toxicity studies carried out over the last decade have reinforced the high acute toxicity to birds of the first-generation neonicotinoids imidacloprid and acetamiprid. A median lethal dose for a sensitive species at the 5 percent tail of a sensitivity distribution (Hazardous dose $_5$ or HD_5) is calculated to be 8 and 10 mg/kg body weight for acetamiprid and imidacloprid respectively. Scaling the acute toxicity to bodyweight to account for the higher sensitivity of small bird species to acute pesticide ingestion yields an estimated HD_5 value as low as 1 mg/kg for imidacloprid. This potentially brings those neonics in the same range as older neurotoxic insecticides viz. organophosphorus and carbamate products. In the decade following our original report, EPA regulators have not amended their faulty assessment of neonic bird toxicity even though they now use distributional methods to assess the toxicity of neonics to other organisms such as aquatic invertebrates. This largely invalidates any EPA effort to carry out a scientifically tenable risk assessment in birds. In this section, we also document doses of incipient mortality or severe physical debilitation. We continue to believe that, preventing large numbers from being debilitated on farm fields is the correct biological and ethical standard against which we should hold pesticides to account. A number of independent studies with imidacloprid have shown that our derived 'debilitation threshold' may still be under-protective with various forms of impairment (e.g. escape responses, disrupted energy budgets, migratory ability etc.) being seen at this dose level or lower.

The acute lethal and chronic toxicity (from industry reproduction tests) of the main neonics was exhaustively reviewed in our 2013 report. A few additional acute toxicity values were found and added to table 1.1 below. Most were generated since our 2013 report was published but a few were missed originally. We have also added the newer neonicotinoid dinotefuran to the table. Table 1.1. separates the toxicity tests into those carried out with either formulated or technical material[3]. Values do not appear to be substantially different. From a risk assessment point of view, it is difficult to determine whether the toxicity to the technical ingredient or the formulated product is most relevant. Soon after an application, it is likely that birds are exposed to formulated material in the field. However, as time passes and weathering occurs, it is likely that the formulation components separate out. Common wisdom had it that liquid formulations were slightly more toxic than active ingredients while the opposite was true for granulars (Hill and Camardese 1984). Therefore, although toxicity values obtained with formulated material are, as a rule, not strictly comparable to those obtained with the technical grade or analytical grade active ingredient, they are undoubtedly relevant, especially in the case of acute exposures soon after application. It has been shown time and time again that a major source of uncertainty in risk assessment comes from interspecies variation in susceptibility. The only way to handle this uncertainty is to look at a larger number of species and use a distribution approach.

[3] The House Sparrow study had mistakenly been included with studies on technical material in our 2013 report

Table 1.1. Acute toxicity of neonic insecticides in birds.

Material	Species	LD_{50} (mg/kg bw)	Probit slope	Dosing method	Reference	Severe signs of incapacitation (mg/kg bw)	Ratio of severe debilitation or first mortality to LD_{50}	Notes
Imidacloprid Technical	Grey Partridge	14.6			Grolleau 1990			
	Japanese Quail	17.0		corn oil	Rawi et al. 2019			Mistakenly reported as a test with formulation in USEPA 2022f. In fact it is with technical material with ≥98% purity.
	Rock Dove	25		gelatin capsule	Grau 1987*	<12.5	<0.50	Value for females. Males 25-50. Severe signs at lowest dose tested – 12.5 mg/kg
	Japanese Quail	31	2.4	gum rabic in water	Grau 1988*	5	0.16	Severe clinical signs at 5 mg/kg. NOEL for clinical signs at 3.1 mg/kg (2.5 mg/kg nominal).
	Canary	35		cremophor EL in water	Grau 1986*	<10	0.29	Midpoint of range (25-50). Serious incapacitation at lowest dose of 10 mg/kg.
	Chicken	50		groundnut oil	Siddiqui 2004			No details given; 6-8 week old leghorn cockerels
	Bobwhite Quail	152	2.7	gelatin capsules	Toll 1990*	50	0.33-0.66	NOEL for clinical signs of 25 mg/kg. Onset of serious incapacitation between 50 and 100 mg/kg.
	Mallard	283	6.6	gelatin capsules	Hancock 1996*	<25	<0.09	Severe signs at lowest dose tested – 25 mg/kg; mortalities up to 8 days post dose. No regurgitation reported (a common issue in Mallard tests) so the value appears to be correct.
Imidacloprid Formulations (commercial or simulated)								

Material	Species	LD_{50} (mg/kg bw)	Probit slope	Dosing method	Reference	Severe signs of incapacitation (mg/kg bw)	Ratio of severe debilitation or first mortality to LD_{50}	Notes
Technical + PEG-600	Japanese Quail	16.0			Rawi et al. 2019			
Confidor 700 g/kg	Japanese Quail	29		water	Ivanova et al. 2013		<0.48	Lowest dose with mortality: 14 mg/kg
Granular 2.5%	House Sparrow	41			Stafford 1991* in CCME 2007, Cox 2001		0.15	Severe debilitation at 6 mg/kg
Confidor 35%	grayish baywing	57			Poliserpi et al. 2021		0.52	Signs of neurotoxicity at 20.6 mg/kg. Severe incapacitation at 29.7 mg/kg and higher. Signs appeared within 30m of dosing, all deaths occurred within 24h of dosing
Imida Nova 60%	Eared Dove	59	6.8	water	Addy-Orduna 2019	<14.1	<0.24	Severe clinical signs seen in lowest dose given
Unnamed formulation 17.8%	Chicken	104			Kammon et al. 2010			Signs of toxicity within 15m of dosing at LD_{50}.
Confidor SL200	Bobwhite Quail	503			EFSA 2014*			
Clothianidin Technical								
	Japanese Quail	423		corn oil	Gallagher and Beavers 2000*	100	0.23	NOEL for clinical signs of 12.5 mg/kg. Light signs at 25 mg/kg. More serious incapacitation at 100 mg/kg.
	Mallard	503			USEPA 2021b			
	Bobwhite Quail	>2000		corn oil	Johnson 1998*	1000	<0.50	NOEL for clinical signs of 500 mg/kg. Serious clinical signs and 20% mortality at both 1000 and 2000 mg/kg

Material	Species	LD_{50} (mg/kg bw)	Probit slope	Dosing method	Reference	Severe signs of incapacitation (mg/kg bw)	Ratio of severe debilitation or first mortality to LD_{50}	Notes
	House Sparrow	528			USEPA 2021b			NOEL for mortality body weight and food consumption of 63 mg/kg. Mortality at 125 mg/kg.
Clothianidin Formulations								
Poncho 60 FS	Eared Dove	4248	1.93	water	Addy-Orduna 2019	392	0.09	Mild clinical signs at lowest dose given (259mg/kg). Severe clinical signs at 392 mg/kg
Thiamethoxam Technical								
	Mallard	576	8.2	methyl cellulose	Johnson 1996*			Emesis observed at all dose levels. NOEL for clinical signs of 137 mg/kg.
	Bobwhite Quail	1552	8.5	methyl cellulose	Johnson 1996*		0.64	NOEL for clinical signs of 500 mg/kg. First mortality, severe debilitation and weight loss at 1000 mg/kg.
Thiamethoxam Formulations								
Cruiser 60FS	Eared Dove	4366	1.49	water	Addy-Orduna 2019	392	0.09	Moderate clinical signs at lowest dose given (66 mg/kg). Severe clinical signs at 392 mg/kg
Actara 250 g/kg	Japanese Quail	4100		water	Ivanova et al. 2013		0.49	Lowest dose with mortality: 2000 mg/kg
Acetamiprid Technical								
	Zebra Finch	5.7	8.6	water	Hubbard 2011*	2.5	0.44-0.63	NOEL for clinical signs of 1.8 mg/kg. Onset of serious debilitation between 2.5 and 3.6 mg/kg.

Material	Species	LD_{50} (mg/kg bw)	Probit slope	Dosing method	Reference	Severe signs of incapacitation (mg/kg bw)	Ratio of severe debilitation or first mortality to LD_{50}	Notes
	Mallard	98	6	sodium carboxy-methyl cellulose	Johnson 1994*	<52	< 0.53	Serious clinical signs seen at lowest dose level of 52 mg/kg.
	Bobwhite Quail	180			European Commission 2004			
Thiacloprid Technical								
	Bobwhite Quail	2716	2.4	Gelatin capsules	Grau 1995*	551	0.20	Clinical signs NOEL of 152. Severe signs onset at 551 mg/kg.
Dinotefuran Technical								
	Zebra Finch	334	6.9		USEPA Pesticide Ecotoxicity Database*			
	Japanese Quail	>2000			USEPA Pesticide Ecotoxicity Database*			
	Bobwhite Quail	>2250			USEPA Pesticide Ecotoxicity Database*			

NOEL = No Observed Effect Level, LD_{50} = Lethal Dose required to kill 50% of a population

* denotes an industry study

1.1.1. A deep dive into the lethality of imidacloprid to birds

Imidacloprid remains the only neonic for which enough data have been generated to mathematically derive a theoretical value for an LD_{50} that will encompass a defined proportion of all exposed species. EPA uses this method now to model the distribution of toxicity values for aquatic organisms exposed to neonics. Why they choose not to do this for birds is a mystery as the number of species now tested allows for this analysis.

To perform the analysis, we used the software developed by EPA (SSD Toolbox). It tests different theoretical distributions of toxicity values (normal, gumbel, triangular, logistic, and Burr) to find the best fit of the extrapolation procedure to existing data through a maximum likelihood method of approximation. Generally, all models with a delta AICc value[4] lower than 2 are considered equally

[4] Akaike Information Criterion for small samples. A non-parametric method to test the plausibility or strength of difference inference models (see Burham and Anderson 1998 for more details).

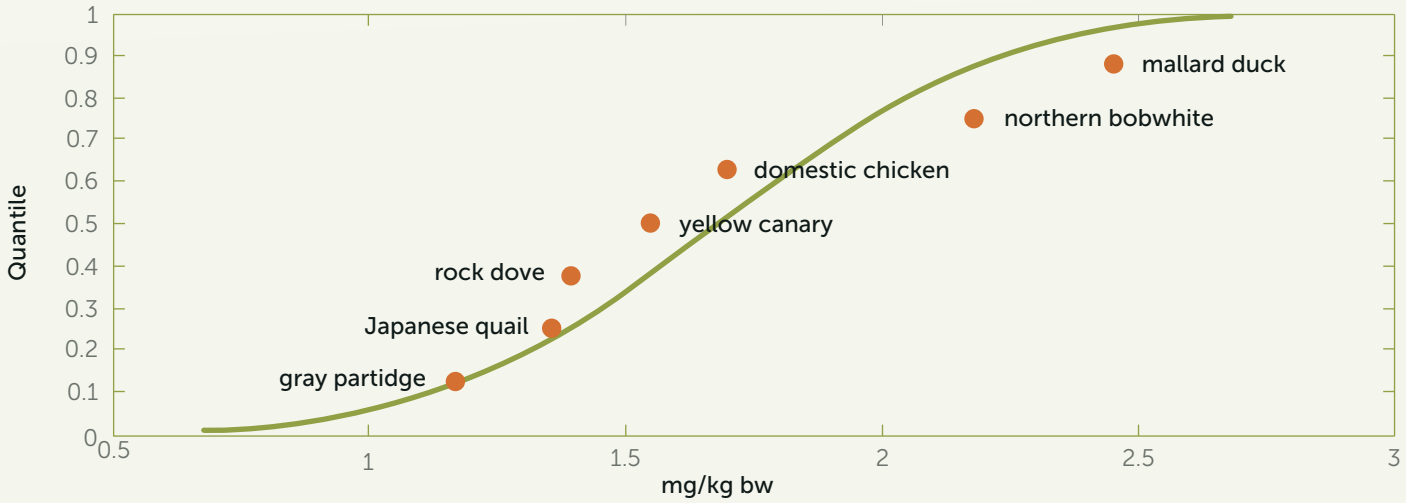
plausible (Burnham and Anderson 1998). For the 7 species[5] tested with technical imidacloprid (see table 1.1.), the estimated HD_5[6] value for the LD_{50} of a bird species at the 5 percent tail of the distribution (more sensitive than 95 percent of other bird species) falls between 7.7 and 13 mg/kg[7] with an average of 10.2 mg/kg for all plausible distributions. In our 2013 report, we considered only the normal distribution model. As seen in table 1.2., a normal distribution may not be the best fit for the observed data. Because of the remaining uncertainty with respect to this (relatively) small number of values, we believe that the most appropriate way forward is to use the average estimate of the HD_5 from available plausible models.

Table 1.2. Model result for raw (uncorrected) LD_{50} data for technical imidacloprid

Distribution	AICc	Delta AICc	Weight	HD_5	SE HD_5
gumbel	79.8765	0	0.3712	13.4892	4.4288
triangular	80.5313	0.6548	0.2676	10.4618	4.0627
normal	81.1698	1.2933	0.1944	9.4075	5.4469
logistic	81.6161	1.7396	0.1556	7.6598	4.8206
burr	86.8783	7.0019	0.0112	13.4839	4.4314

The following figure (Fig. 1.1.) shows the cumulative distribution of the available acute toxicity data for technical imidacloprid in birds.

Figure 1.1. Cumulative probability distribution of acute toxicity values for technical imidacloprid



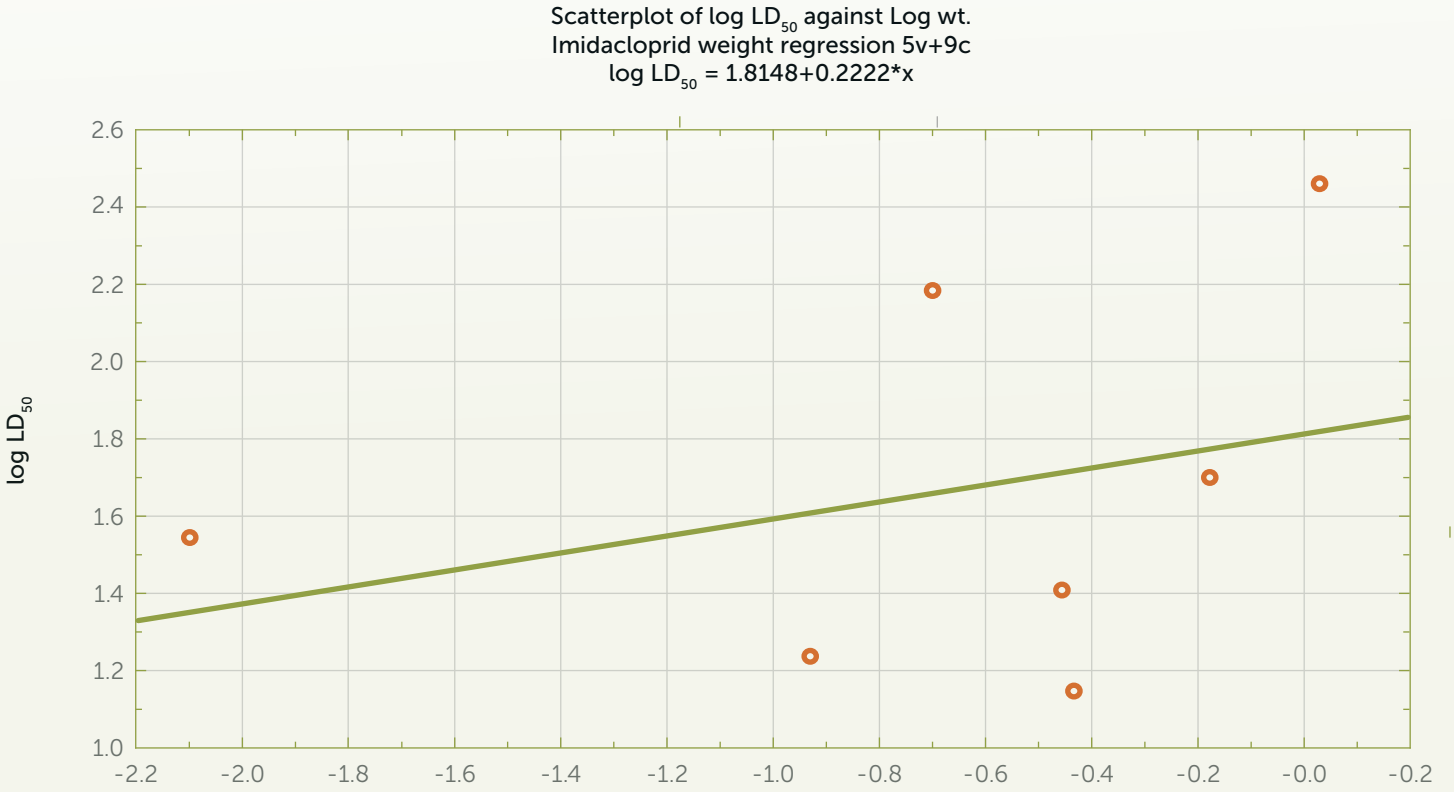
[5] For Japanese Quail, the geometric mean of the two available values is used.

[6] 'Hazardous dose' to 5% of the population

[7] Toxicity values are expressed as mg/kg body weight of the test organism

However, Mineau et al. (1996a) observed that small birds are generally more sensitive than larger ones to a wide range of pesticides even when LD_{50} values are expressed as mg/kg body weight. Inherent in the concept of toxicity values being expressed as mg/kg bw is the assumption that toxicity scales perfectly to weight; that is, toxicity expressed as mg/bird would scale to weight with a slope of 1. As argued by Mineau et al. (1996a) and accepted by the USEPA in their risk assessment procedure (e.g. T-Rex 1.5 – USEPA 2023a), it is reasonable to use the empirical value observed in the dataset being analysed, whether or not the observed slope is significant or different from one. The observed LD_{50} values for technical imidacloprid (expressed as mg/kg bw) are plotted against weight in figure 1.2 below. The data are given in table 1.3. The observed slope of the relationship is 1.81.[8] This is quite an extreme scaling value compared to most known compounds (Mineau et al. 1996a). Methiocarb, a carbamate compound used as a bird repellent because of its ability to quickly elicit a conditioned aversion response in birds had a well-defined slope of 1.4 (based on 32 species being tested). Because the Mallard is the most insensitive species tested and because of the propensity of this species to regurgitate part of its dose, we double-checked the original study (Hancock 1996) which we had obtained from a Freedom of Information request for our 2013 report. The report describes signs of toxicity in detail but makes no mention of regurgitation in any of the birds tested.

Figure 1.3. Log-log plot of LD_{50}values against body weight for acute imidacloprid data in birds.



Log wt.:log LD_{50}: y= 1.8148 + 0.2222*x; r = 0.3189, p = 0.4587

[8] This is the slope when LD_{50} values are expressed as mg/bird; 0.81 when expressed as mg/kg. TRex recommends a value of 1.15 (0.15) when no slope term is available.

The reason behind this very real phenomenon has never been completely elucidated[9] although the most plausible reason is that small birds tend to be more sensitive to the physical disruption associated with an acute toxicity test, primarily an inability to feed and survive on its body reserves. A small bird that is unable to feed itself properly may not survive the night following dosing but a fat lab-reared Mallard (*Anas platyrynchos*) or Bobwhite Quail (*Colinus virginianus*) can probably go for a few days without food and still survive, thereby increasing the apparent LD_{50} value. For example, in the original Mallard acute toxicity test (Hancock 1996), one of the dosed birds is noted as having survived seven days post-dose without any food consumption. The ability of birds to survive a dose that should be sub-lethal from a pure toxicological basis is, of course, highly relevant to a field situation where contaminated individuals are exposed to the elements, predators, conspecific competitors, etc. The very wide difference between the toxicity of acetamiprid and dinotefuran to Zebra Finches (*Taeniopygia castanotis)* compared to the usual test species (Bobwhite Quail and Mallard) suggests that this scaling is very much a factor with neonicotinoids. Smaller birds such as Zebra Finches are seldom tested although this is becoming more frequent.

EPA recommends the following when carrying out a risk assessment for birds or mammals:

Excerpt from USEPA's TRex 1.5 [10]

- Adjusted avian LD_{50}

  $Adj.\ LD_{50} = LD_{50}\,(AW\ /\ TW)^{(x-1)}$

  where:

  $Adj.\ LD_{50}$ = adjusted LD_{50} (mg/kg-bw) calculated by the equation

  LD_{50} = endpoint reported from bird study (mg/kg-bw)

  TW =bodyweight or tested animal (178g bobwhite; 1580g mallard)

  AW = bodyweight of assessed animal (avian: 20g, 100g, and 1000g)

  x = Mineau scaling factor for birds; EFED default 1.15

In order to be consistent with our previous report and with a published compendium of HD_5 values in birds (Mineau et al. 2001), table 1.3 shows LD_{50} values corrected for a small (20g) bird – such as a finch or sparrow. Here, we show the results showing both the empirical scaling factor of 1.81 and the USEPA default value of 1.15.

[9] If the slope term was based on physiological principles of scale such as metabolic or clearance rates, it would likely be less than 1. This is clearly not the case for birds and a large number of pesticides examined to date (Mineau et al. 1996, 2000).

[10] It should be noted that the EPA-recommended weights for northern Bobwhite Quail and Mallard especially differ from the ones we introduced in our 2013 report and are still using here. The larger body weight used by EPA will tend to increase the slope and enhance the effect of scaling.

Table 1.3 Measured and weight corrected LD_{50} values for imidacloprid

Genus Species	Measured LD_{50} (mg/kg)	Wt.(kg)	LD_{50} corrected for 20g bird (mg/kg) with 1.81 scaling factor	LD_{50} corrected for 20g bird (mg/kg) with 1.15 scaling factor
Grey Partridge	14.6	0.37	1.4	9.4
Japanese Quail [a]	23	0.12	5.4	17.6
Rock Dove	25	0.355	2.4	16.2
Canary	35	0.008	73	40.2
Domestic Chicken	50	0.672	2.9	29.5
Northern Bobwhite Quail	152	0.2	23	108
Mallard duck	283	1.082	11	155
HD_5 value (mean of all plausible models)	10.2		0.97	7.7

[a] Where several values are available for the same species, a geometric mean of the values is used.

In its most recent assessment, USEPA (2022f) uses 17 mg/kg as the endpoint with which they carry out their assessment for endangered species. This is merely the lower value of the studies they have in hand. There is no recognition on their part that it is highly unlikely the most sensitive bird species has been found from such a small number of tests. Principles of species sensitivity distributions which they now apply to aquatic invertebrates are not being applied in the case of vertebrate species. There is no indication as to why this is the case. The lower value of 14.6 mg/kg obtained by a research institute of the French government and communicated to European and Canadian authorities (Grolleau 1990) was presumably not submitted to the EPA.

As table 1.3 above shows, there are legitimate grounds to question the validity of USEPA's risk assessment of acute effects in birds. For imidacloprid, it is likely that a sensitive species, whether an endangered species or not, will have an LD_{50} much lower than the 17 mg/kg value they propose. As shown above, simple principles of species sensitivity distribution on the available raw data without any scaling correction return an estimate of **10.2 mg/kg**. Given the higher apparent sensitivity of birds of smaller mass, running the species sensitivity distribution using the same methods with LD_{50} values corrected for a small (20g) species returns a value of **7.7 mg/kg** using the USEPA default scaling value of 1.15 or the extremely low (and disturbing) value of **0.97 mg/kg** using the observed scaling relationship of 1.81. In short, there is enough uncertainty to question the current EPA belief of the imidacloprid acute toxicity in birds.

1.1.2. Comparative lethality of neonics to birds

As reviewed in 2013, most of the difficulty in ascertaining the toxicity of pesticides to various organisms is the wide interspecies differences that are always present. Where sufficient data exist, the most robust way of setting a toxicity endpoint is through the use of species sensitivity distribution as described in section 1.1.1. Where the data are insufficient, mathematical methods have been devised to approximate an HD_5 based on the average degree of variation seen in birds with other pesticides (Aldenberg and

Luttik 2002). Another approach is that of Mineau et al. (2001); deriving extrapolation factors from the usual test species (Bobwhite Quail and Mallard) to include all bird species while considering the scaling of pesticide toxicity to body mass explained in section 1.1.1.

Table 1.4. summarizes the levels found to lead to lethality and debilitation in test species to date. All values are reported as mg of active substance per kg of body weight and arranged from most to least toxic. See the original report for details and references.

Table 1.4. Summary of actual and derived acute toxicity test endpoints for all neonic insecticides.

Compound	Min. recorded LD_{50}	Max recorded LD_{50}	Median values LD_{50}	Unscaled HD_5	HD_5 scaled to a 20g species [b]	Min. recorded dose showing clear debilitation or first mortality
acetamiprid	5.7	180	98	8	20.9	3.0
imidacloprid	14.6	283	35	10.2	0.97 – 7.7 [c]	5.0
Dinotefuran [a]	331	> 2250	2000	197	263	-
Clothianidin [a]	430	> 2000	515	118	84	100
thiamethoxam	576	1552	1064	162	98	> 137
thiacloprid	2716	2716	2716	467	315	< 551

[a] Accepting > limit values as true estimates

[b] All compounds except imidacloprid based on small sample estimations; Aldengerg and Luttik 2001 (unscaled); Mineau et al. 2001 (scaled)

[c] See section 1.1.1. above

The degree of confidence we have varies a great deal from compound to compound depending on the extent of testing. The HD_5 is one attempt used by a number of regulatory agencies to reduce the impact of different testing intensities. The following observations can be made:

- We should be more concerned about acute toxicity to birds in the case of acetamiprid and imidacloprid and less so in the case of clothianidin and thiamethoxam. The acute toxicity of clothianidin and thiamethoxam to birds clearly appears to be less. Whether or not clothianidin and thiamethoxam present an acute risk following exposure will, of course, depend on the magnitude of exposure. As reviewed before, seed treatments represent a large potential for a very large exposure in a very short amount of time.

- At first blush, thiacloprid appears to be less acutely toxic to birds also but this conclusion should be tempered by the fact that only one data point exists; and by the fact that thiacloprid appears to be the most toxic of the five in mammals (Mineau and Callaghan 2018). Clearly more testing is required to conclude one way or the other. The relative rank of thiacloprid is all the more uncertain given the level at which the 'official' avian reproductive studies present a risk. As detailed in our 2013 report, thiacloprid appears to group with imidacloprid and acetamiprid in showing a higher reproductive toxicity to birds (see table 2.1 below).

In 2013, we commented on the fact that, with neonicotinoids, severe signs of toxicity were seen in tests at doses as low as 1/10 of the reported lethal toxicity. For example, for Mallard duck, the least sensitive bird species tested with imidacloprid registered an LD_{50} value of 283 mg/kg. However, severe signs of toxicity were seen at the lowest dose given; 25 mg/kg (table 1.1). Signs of severe debilitation seen in laboratory situations have real world relevance as they suggest an impact on birds in the wild ingesting or being exposed to a pesticide, even if the exposure is not initially lethal. This is an important point to remember since most formal risk assessments use the calculated median lethal toxicity as the endpoint against which exposure levels are compared. This is clearly under-protective from many points of view; especially so for wild birds that do not have a warm, undisturbed laboratory environment to recover from a pesticide injury. We propose to use the concept of incipient mortality/debilitation in our risk assessment in this report (see section 1.4 below). Protection of individuals rather than protection of half of the population of exposed species is paramount when considering a likely harm to rare and endangered species. As a matter of policy, we believe that the appropriate standard of protection in our farm fields, whether from an ecological or ethical point of view, is not to have millions of birds convulsing or being paralyzed from their exposure to pesticide residues.

1.2. Sub-lethal and chronic toxicity of neonics to birds

1.2. Neonics, especially the high-volume compounds imidacloprid, clothianidin and thiamethoxam, do not show the extent of differentiation among compounds for chronic and reproductive toxicity in birds as they do for acute toxicity. The original industry studies extensively reviewed in our 2013 report have not changed; new independent studies, however, give credence to our derivation of a reproductive benchmark in the form of an 'MATC' (the geometric mean of no-effect and low effect levels expressed here as mg/kg bodyweight/day) extrapolated to a reasonably sensitive bird (one at the 5 percent tail of an acute toxicity sensitivity distribution). In particular, several new studies with imidacloprid have shown effect levels at very similar or lower doses. Based on recent studies, we have increasing concerns over reproductive and sub lethal effects resulting from low exposures in farm fields. In particular, impacts on sperm quality have been seen at dose levels a fraction of our calculated MATC; viz. 0.05 percent of the reproductive threshold for acetamiprid; 1.1 percent of the reproductive threshold for clothianidin. Given that exposure is often season-long, this raises the spectre of significant effects on a large number of bird species. Functional impairment post-dose has now been documented under various conditions and in different species; these findings will be used in the risk assessment reported below (section 1.4).

When new pesticide products appear, standard toxicity test are often the only information available; but as time passes by, more and more testing occurs, often in different university laboratories and with different protocols and endpoints. Neonics are no exception and some useful information has emerged in the last 10 years.

1.2.1. Effects on Reproduction

Our 2013 report reported the following levels of chronic toxicity obtained through the standard laboratory reproductive tests on Bobwhite Quail and Mallard. Although these are tests of reproductive potential, the test protocol examines a very truncated portion of the reproductive process and is best seen as a more generic test of chronic toxicity (Mineau et al. 1994, 1996b). The values given here extrapolate the results obtained in the Bobwhite Quail and Mallard to a sensitive bird species (in the lower 5 percent tail of the distribution) based on the degree of variation in acute toxicity seen among tested species (table 1.5). The logic here is that it is highly unlikely for sub-lethal toxicity in all of its complex manifestations to be less variable among species than lethal toxicity as shown by

a simple LD_{50} test (Luttik et al. 2005). That reference as well as the original 2013 report (Mineau and Palmer 2013) should be consulted for a detailed description of the method and the individual study references. Two corrected values are given for low effect levels with clothianidin and thiacloprid. These were estimated based on an average degree of separation between NOAEL and LOAEL[11] values rather than the more stringent criterion used in our 2013 report. Dinotefuran has been added to the table. However, we did not assess the original reproductive studies with dinotefuran as we had for the previous compounds[12]. Endpoints were accepted as reported by the USEPA in their ecotoxicology database (https://cfpub.epa.gov/ecotox/index.cfm).

The standard avian reproduction test typically generates a NOAEL and LOAEL. Despite the fact that the NOAEL is often not a true NOAEL because the test has an insufficient power/sample size to detect more subtle effects, it is common practice (in aquatic toxicology at least) to derive the geometric mean of those two points. It is termed MATC or **M**aximum **A**cceptable **T**oxicant **C**oncentration. We have chosen to use this concept here so as not to be accused of being too protective in the interpretation of industry-generated reproduction tests. Of course, our MATC is no longer a 'concentration' as the acronym implies but a 'dose level' calculated in mg/kg of body weight for the test birds given the laboratory food concentrations that were used in the tests.

As pointed out in the previous chapter, this meager evidence tends to place the toxicity of thiacloprid much higher than suggested from the acute data. There is far less separation of the different active ingredients with the exception of dinotefuran which appears to be a full order of magnitude less toxic in the standard reproduction tests.

Table 1.5. Avian chronic toxicity data extrapolated from industry tests on Bobwhite Quail and Mallard.

Active ingredient	Avian Chronic - Critical intake level (NOAEL - mg/kg/day) for a sensitive bird at 5% tail of acute sensitivity distribution	Avian Chronic - Critical intake level (LOAEL - mg/kg/day) for a sensitive bird at 5% tail of acute sensitivity distribution	MATC. Geometric mean of the NOAEL and LOAEL (mg/kg/day)
Imidacloprid	1.41	2.82	1.99
Thiacloprid	1.61	4.26	2.62
Acetamiprid	2.49	4.97	3.52
Thiamethoxam	4.22	12.66	7.31
Clothianidin	5.10	14.8	8.69
Dinotefuran	47.3	199	97.0

[11] No Observed Adverse Effect Level (NOAEL often shortened to NOEL given that observed effects are usually adverse); and Low Observed Adverse Effect Level (LOAEL – the lowest dose at which effects are seen in the study under consideration).

[12] Industry studies used in our 2013 report were obtained through a freedom of information request.

It should be mentioned here that scientists within EPA (Etterson et al. 2017), following up on recommendations formulated by an expert group assembled by Britain's Department for Environment, Food and Rural Affairs (DEFRA) more than a decade earlier (Roelofs et al. 2005)[13], developed an innovative way of modeling results obtained from industry reproduction studies in birds. However, we see no indication that any of the EPA assessments on neonicotinoids have used these methods.

In our previous report, we briefly reviewed the then-breaking research of Lopez-Antia et al. (2013) with imidacloprid seed treatments and that of Tokumoto et al. (2013) with formulated clothianidin. The latter, especially, had raised new concerns (the authors were concerned about the critically endangered Crested Ibis (*Nipponia nippon*) ingesting treated rice seed) with a dose-related increase in testicular abnormalities and DNA breakage in germ cells of Japanese Quail (*Coturnix japonica*) even though fertility was not significantly affected. The mechanism for this effect on germ cells was thought to be operating through oxidative stress. Embryo length was reduced also in a dose-dependent fashion with significant adverse effects seen in the 1 mg/kg/day (administered for a 30-day period) group. Working in the same laboratory, Hoshi et al. (2014) reported on a further experiment where quail were dosed at 0.01, 0.1, 1 and 10 mg/kg clothianidin daily for a 6-week period. In male birds, there was a clear dose response in the number of aberrant (single stranded) DNA in the bird's testes; this was already visible at the lowest 0.01 mg/kg dose and statistically significant at the 0.1 mg/kg level. In females, they saw a decline in egg-laying rates starting at 1 mg/kg but statistically significant at 10 mg/kg only. Unfortunately, the use of formulated material means that the relative effects of clothianidin and that of the other formulation components could not be formally separated although the similarity to effects seen in mammalian system (*i.e.* oxidative stress) strongly suggest that the active ingredient clothianidin was responsible for the effects seen.

Continuing with their work on Grey Partridges (*Perdix perdix*) exposed to imidacloprid-coated wheat seeds, Lopez-Antia et al. (2015) found that birds consuming an estimated 8.1-12.6 mg/kg bw /day started dying on the third day of exposure. Birds given seeds treated with 1/5 of the concentration (1.6 – 2.5 mg/kg bird/day) were able to breed but took longer to produce eggs, had reduced clutch sizes and the chicks showed a reduced immune response. Unfortunately, the experiment did not separate the effect of the reduced food consumption (from the second day on test onwards) from the effects of the chemical.[14] Nevertheless, this dosing level is very similar to the reproductive effect level we calculated based on Mallard and Bobwhite Quail tests for a reasonably sensitive species, namely 2.8 mg/kg/day. The Grey Partridge had already shown to be a comparatively sensitive species in acute tests (table 1.1).

Several neonics have been found to have endocrine effects when tested in mammals (reviewed in Mineau and Callaghan 2018). Pandey and Mohanty (2015, 2017) and Pandey et al. (2017) dosed male Red Munia, small passerines from the Indian sub-continent (8.5g – *Amandava amandava*) at 0.77 mg/kg/day and 1.5 mg/kg/day of formulated imidacloprid[15]. The pesticide was mixed in the bird's food and given over a 30-day period during the pre-breeding and breeding season. In a series of experiments, these authors documented clear disruptions to the bird's hypothalamic-pituitary-thyroid (HPT) axis at both doses with birds showing hypothyroidism, as well as a reduction in breeding readiness and behaviors, loss of breeding plumage, and effects on testicular development and plasma testosterone levels. Singing started decreasing one week after the start of the experiment and stopped completely

[13] Full disclosure: The senior author of this report (PM) was closely involved with the original work and has been following its uptake (or lack of) since then.

[14] The way to do this is to have a pair-fed group of bird who are given the same amount of untreated feed.

[15] The authors described this as 0.25% and 0.50% of a reference dose of 31 mg/kg/day.

by the 4th week. Various effects were found to be synergised with the addition of the dithiocarbamate fungicide mancozeb; and the fungicide alone also produced endocrine disruptions.

Humann-Guilleminot et al. (2019b) review the extensive body of evidence linking neonicotinoid exposure to male fertility and reproductive ability in both birds and mammals. Their own study involved dosing House Sparrows (*Passer domesticus*) with acetamiprid. This neonicotinoid was not included in the 2018 neonic ban in Europe. Yet, as seen in tables 1.1 and 1.4 above, it may even be more toxic to birds than imidacloprid. They gave each bird the amazingly low dose of 0.03 mg/kg bw divided into 7 equal doses over 19 days, capturing at least one full spermatogenic cycle. That very low dose did not result in any body mass loss in the birds. Sperm quality (measured as density) was reduced by 52%, raising the possibility that such a minute dose (0.05 percent of our derived MATC reproductive threshold) of acetamiprid could affect fertility of exposed birds. The effect was not thought to be through free-radical effects as postulated in the other studies reviewed above.

In an unpublished MSc thesis, Sundall (2020) studied the effects of clothianidin-treated corn seed on the breeding success of Ring-necked Pheasants. Birds were gavage fed with differing numbers of treated seed for either 14 days or 30 days in two separate experiments. Eggs were collected and incubated as they are in industry reproduction studies. Breeding birds given 75 treated seeds per day showed substantial mortality (the LD_{50} was estimated to be 77 seeds per day). However, no weight loss was seen in the surviving birds – therefore no anorexic response was induced by the gavage as seen in many other captive studies. Balance problems and convulsions were occasionally observed in the hens given 15 treated seeds/day or roughly 8.4 mg/kg/day given the average weight of the birds. Based on the author's calculations, signs of severe intoxication were therefore seen at roughly 20% of a median lethal dose given over 14 days. Effects on chick growth were seen in the higher dose group of 75 seeds/bird/day or 42 mg/kg/day.

Zgirski et al. (2021) dosed Zebra Finch nestlings with 0.205 mg/kg/day of imidacloprid from day 4 to 30 after hatching. Surprisingly, this amount of dosing increased growth rate (body condition rather than structural body size) and ameliorated the food restriction imposed on some of the nestlings. When allowed to grow to maturity, birds at 90 and 800 days of age still showed differences with the imidacloprid-treated birds showing higher scaled lean body masses and higher fat levels. The authors commented that these changes from the evolutionarily-determined ideal weight were not necessarily favourable as they would entail higher maintenance costs in adulthood. The authors concluded that at very low doses, imidacloprid may show a hormetic response. Thyroid hormones were apparently not involved in this process as has been the case in other research, both avian and mammalian.

In reviewing recent findings on the reproductive toxicity of neonics to birds, we did not review studies using egg injections to look at the potential of the chemicals to cause physiological, biochemical or developmental effects (e.g. Hussein et al. 2014, Wang et al. 2016). It is difficult to judge the pertinence of these findings in a real-world context. This review remains focused on the very real and present dangers posed by neonics to birds whether direct effects through exposure in farm fields following standard agricultural practices or indirect effects through loss of feeding opportunities.

1.2.2. Sub-lethal Effects other than Reproduction

The same investigators who established an acute lethal dose of 104 mg/kg for formulated imidacloprid in Chickens found that giving 1/20 of the lethal dose (or 20 mg a.i./kg) for 45 days led to immunosuppressive effects (Kammon et al. 2012a). These effects could be reduced by simultaneous dosing with antioxidants (Kammon et al. 2012b).

Studies with neonicotinoids other than imidacloprid are quite a bit rarer. In cockerels (Chickens – *Gallus domesticus*) daily administration of 250 mg/kg thiamethoxam for 15 days resulted in significant variations in a large number of blood markers (Gul et al. 2017). It is not clear whether the material was in active ingredient form or formulated material.

Highly relevant deficits were elicited in birds exposed to imidacloprid (Eng et al. 2017). White-crowned Sparrows (*Zonotrichia leucophrys*) dosed by gavage at 4.1 mg/kg bw showed symptoms of ataxia, and lethargy, reduced their food consumption, and lost body mass. Following three consecutive days of dosing, they could not properly orient for migration in a funnel test where birds are exposed to night skies and their desired direction of departure is established. Normal birds dosed with the oil vehicle only did orient as expected for the time of year and normal migratory direction. It is not completely clear whether the orientation ability deficit was a direct effect of imidacloprid or was in part a result of reduced food consumption and weight loss – both of which can affect performance. However, similar deficits on migratory orientation where seen in birds dosed with the cholinesterase-inhibiting organophosphorus insecticide chlorpyrifos although these birds showed no weight loss at the dose given. This strongly suggests that the deficits in orientation ability were caused by direct effects on the cholinergic system, both insecticides affecting these neural pathways, albeit in different mechanistic ways. Birds dosed with imidacloprid had recovered their ability to orient at some point between 3-14 days after dosing; birds given chlorpyrifos had not recovered by the 14-day end of test.

In a ground-breaking follow up to this study, Eng et al. (2019) captured and dosed southbound migrating white crowned sparrows, assessed the time it took them to leave the area post dose and then followed their movements over several hundred kilometers. Unlike what was found in the laboratory, dosed birds, once they migrated did so in the right direction and with normal flight speeds. However, dosing gave rise to a loss of fat reserves and, extended the length of their migratory stopover. Pesticide-induced anorexia was a common feature of cholinesterase-inhibiting compounds such as organophosphorus insecticides (*e.g.* Mineau et al. 1990) and is clearly a factor with neonicotinoids also. As pointed out by Eng et al. (2019), a reduced ability of birds to accumulate fat reserves could have profound consequences in real life. Indeed, the same argument could be made with any wildlife species exposed to neonicotinoids. They documented a delay in migration but the list could extend to increased risk of predation, disruption in breeding ability, inability to feed a brood etc. Given that neonicotinoids affect the cholinergic system, it is useful to look at the wide potential of direct effects possible with a disruption of these critical neural pathways (e.g. Mineau 1991, Grue et al. 1991). Eng et al. (2019) found clear effects after a single dose of imidacloprid as low as 1.2 mg/kg bw. Although a single acute dose, it is noteworthy that this value is below our reproductive no-effect level of 1.4 mg/kg/day estimated for a bird at the lower 5 percent tail of species sensitivity (table 1.5).

Bean et al. (2019) reported that startle and escape responses as well as muscle strength were affected in Japanese Quail at doses of 3 percent of the imidacloprid LD_{50} (taken to be 31 mg/kg based on another study); therefore 0.93 mg/kg. This is about 1/3 of our proposed debilitation threshold. However, in a different experiment, individuals receiving up to 95 percent of this LD_{50} appeared normal to the observers. This raises an interesting point that applies to much of toxicology research, whether with neonics or other pesticides. Animals are heavily selected to appear 'normal' since weakened, disabled or erratically-behaving individuals can be expected to be taken quickly by predators or possibly, attacked by conspecifics. Seeing clear signs of distress in dosed birds is an indication that these individuals are highly compromised and that some form of functional impairment probably occurred at a much lower dose.

Franzen-Klein et al. (2020) dosed young (6-9 week old) Chickens with imidacloprid at doses ranging from 0.03 mg/kg/day to 15.5 mg/kg/day for 7 consecutive days. All birds were seen to recover from

one dose to the next and no cumulative toxicity was evident. The authors therefore refer to their study as a repeat acute dose study rather than a chronic dosing experiment. Unlike many standard tests where dosed birds are fasted, the Chickens in this work were not fasted and, furthermore, the doses were mixed with a grain formula for chicks. This should be a best case situation for the birds and one most closely mimicking dosing in the wild. Fasted birds typically absorb more of the material with which they are being dosed. They established an effective dose (ED_{50}) of 4.6 mg/kg/day for any neurobehavioral effects (ED_{10}= 2.2 mg/kg) and 11.2 mg/kg/day (ED_{10} = 2.5 mg/kg) for more severe and longer-lasting effects. At the doses given, they did not see any immune effects. A key part of this work is that they measured the exact time for neurobehavioral deficits to be seen after dosing and the duration of symptoms – two key parameters to assess the risk of birds in the field. The most sensitive bird recorded in this work developed slight neurobehavioral impairment at 0.34 mg/kg. At the upper end of dosing, only two individuals on separate days appeared normal following 15.5 mg/kg. Signs of toxicity appeared after the first and second dose avg: 65 +/- sd 49.5 min. and lasted 45 min. on average. The most rapid onset of signs was 3 min. post dose in two individuals dosed either with 3.42 mg/kg or 10.25 mg/kg. Severely affected individuals showed major impairment lasting 5.5 h. Weight loss was only seen in the highest 15.5 mg/kg/day group.

We have long argued that the type of information obtained by Franzen-Klein et al. (2020) should be required for anyone carrying toxicity tests. The information on onset and duration of signs is critical in assessing their real-world importance. To not provide detailed observations of onset and duration is a serious missed opportunity (Callaghan and Mineau 2008).

English et al. (2021) placed male ruby-throated hummingbirds (*Archilochus colubris*) hummingbirds dosed with imidacloprid in respirometers to assess their energy expenditure 1.5 - 2 h after dosing. Their flight and foraging behaviour was assessed 4h post dose. Birds were dosed through contaminated sugar solution and received either 1, 2 or 2.5 mg/kg bw imidacloprid. The doses were chosen to be environmentally relevant given the nectar concentrations of imidacloprid measured by the authors in the course of another study around blueberry farms. The results showed that the birds reduced their energy expenditure in a dose-related fashion with an average 6 percent reduction in activity evident at 1.0 mg/kg. It is clear that even this small dose would interfere with normal foraging activity for the first hours post dose. Effects had declined by 2.5 – 3.5 h post dose. Although this may seem like a short impairment time, small birds such as hummingbirds live on a precarious energetic knife edge and we would expect even short impairment periods to have consequences on breeding or survival. No effects on immune response were seen at these low doses.

1.3. The increasing evidence of real-life exposure and impacts in the field

1.3. Neonics are not repellant chemicals. Birds may avoid lethal poisoning if 1) their rate of intake of contaminated food items is low enough that they will be made too sick to eat before ingesting a lethal dose; 2) they can recognize what food item has sickened them so that it can be avoided; 3) there are no other 'hidden' sources of exposure beyond their direct control such as drinking water, dermal contact of inhalation of fine droplets or vapors. Learned avoidance in laboratory settings has been found to be highly variable and dependant on test conditions. This is not a new finding. Similar results were seen with older seed products such as organophosphorus insecticides. Learned avoidance is not foolproof and there is often mortality even when the population response shows avoidance. In order for learned avoidance to work, birds have to be made sick and anorexic ... often for several hours or up to a day post exposure. Given the scale of use of neonics, is it acceptable to have millions of birds being sickened, made anorexic, paralyzed or exhibiting convulsions following exposure? How many of those birds will appear to leave the fields unscathed to seek cover and die in the following days.

This scenario, which was all too common with cholinesterase-inhibiting compounds of medium toxicity is a probable one here, at least with the first generation neonics.

Treated seeds remain an easy route of exposure for birds. Indeed, many farmland species have become dependent on seeds. Regardless of the seeding technology, there are always small spills of concentrated surface seeds available to birds. This is standard farming practice and it is cynical to think that this can be avoided. The answer is not to blame or criminalize the farmer but rather to ensure that seed treatments are safe for birds and other wildlife.

Observational studies (at least those not carried out by the manufacturers) show that many species are attracted to and will feed on treated seeds, especially when available at spills. Numerous kills have been reported with imidacloprid in Europe, North America and elsewhere.

Residue analyses have confirmed that exposure of birds to neonics are commonplace. Although they are rapidly cleared from tissues in vertebrates, surveys of a broad range of species show that a large proportion of most species examined show evidence of exposure. Gamebirds and seed eaters frequenting farmland are clearly massively exposed; however, residues are being seen in raptors, hummingbirds and even seabirds. Neonics may not bioaccumulate in organisms and biomagnify in food webs as did DDT and other organochlorine pesticides of old, but they appear to be as widely distributed in the broader terrestrial and aquatic environments.

In 2013, we reviewed industry claims that neonic-coated seeds would not be taken by birds on account of the seeds being 'repellant'. As discussed in that earlier report, there is a major difference between a compound that is a true sensory repellant – i.e. birds immediately spitting out or rejecting a treated seed on sight – versus conditioned aversion where birds learn that a certain food type is toxic when they feel the onset of toxicosis. Unless tests are properly designed to differentiate between those two scenarios, it is easy to delude oneself – or, more pertinently, delude regulators – in thinking that pesticide-treated seeds are repellant and therefore should prove safe under real-world conditions.

A number of studies on imidacloprid and clothianidin, many of them from industry and claiming repellency in birds, were reviewed in our 2013 report (Mineau and Palmer 2013; section 5.2). We concluded, like EPA regulators before us (USEPA 2007 for example), that any avoidance was clearly learned and mediated by post-ingestion distress. In other words, the birds have to be made sick before they slow down their rates of food intake. The key question then is whether a bird's first meal of treated seed is going to cause irreparable harm (debilitation or death) before the bird realizes that the food in question should be avoided.

1.3.1. Laboratory approaches to looking at conditioned aversion

The data from Lopez-Antia et al. (2014) suggests that it took a day for Grey Partridge to recognize the toxic nature of imidacloprid-treated seed; at least, consumption of seed on the first day of test was not significantly different whether the birds were given untreated seed vs. imidacloprid-treated seed. Based on other studies reviewed below, as well as Franzen-Klein et al. (2020) reviewed earlier, this appears to be an extreme value.

It has been shown that, the more difficult it is for birds to associate treated seeds with post-ingestion illness, birds will consume more of the treated seed. Lopez-Antia et al. (2014) found that the ingestion of imidacloprid-treated seed by captive Grey Partridge depended on how food was presented. Having the treated seeds offered in one dish, even if the location was moved relative to a dish of untreated

seed was not sufficient to 'fool' the birds and they managed to avoid the treated seed. However, increasing the number of feeders in which the treated seed was distributed increased the amount of treated seed ingested. It has long been known that it is difficult to replicate wild feeding opportunities in a laboratory environment. Changing the shape of the dish, introducing periods of hunger as well as competition for conspecific foragers all can influence the results (Pascual et al. 1999). There are well documented cases where compounds shown to be well avoided in the laboratory (e.g. diazinon applied to turfgrass) proved disastrous in real life (Mineau et al. 1994).

In an unpublished MSc thesis, Sundall (2020) gave captive Ring-necked Pheasants a three-way choice of untreated, red-dyed seed with no insecticide or red-dyed (Rhodamine B) corn seed treated with clothianidin at the rate of 1.25 mg a.i./seed. Untreated seeds were clearly preferred over the course of the 7 day study. The groups of captive birds were said to be 'slow to commit' to one of the seed types initially and one group of birds developed a preference for the Rhodamine-B treated seeds.

In a recent article, Addy-Orduna et al. (2022) looked at the establishment of imidacloprid, clothianidin and thiamethoxam-induced conditioned aversion in captive Eared Doves. Their study protocol consisted of providing untreated test seed (sorghum, soy, soy cotyledons) for 5-7 days for one hour each morning (maintenance food was then offered 21/2 hours later), a 3-5 day exposure to the same food items but treated with imidacloprid, a rest period of 7-8 days, a second 3-5 day exposure to contaminated food, and finally, a 3-5 day exposure to untreated food. Some birds were videotaped for the first 15 minutes following food presentation. Birds exposed to the neonic showed both a conditioned aversion and an anorexic response for the first few days following exposure. Thereafter, they increased their maintenance food consumption to make up the deficit. The birds did not form a conditioned aversion to the test seed types and ate similar amounts of those foods as the control birds when the foods were presented untreated. Videotape analysis showed that there were differences in pecking rates tallied for the first 15 minutes of presentation. Unfortunately, there was no attempt to distinguish the onset of the conditioned response on a finer time scale.

Despite the formation of a strong conditioned aversion response, it was not enough to prevent mortality. Three of eight birds died with imidacloprid, two on the first day of exposure, one on the second. The average liver residue was 5.66 µg/g. Signs of toxicity were observed with the three sorghum seed treatments under study (imidacloprid at 4.5 g a.i./kg seed; clothianidin at unknown concentration; thiamethoxam at 1.9 g a.i./kg seed).[16] Results were more difficult to interpret with the soybean because they did not prove to be a popular food item whether treated or not. They concluded that soy would not likely prove a risk to this species. With clothianidin-treated seed, one of eight birds died on the second day of exposure. All three seed-treatment chemicals gave rise to a similar degree of conditioned aversion despite their very different acute toxicities. Indeed, conditioned aversion was said to be the strongest with clothianidin. That conditioned aversion was not transferred to untreated seed suggests that treated seed were recognized visually. The authors reviewed their results in the context of industry conditioned aversion studies, many of which were reviewed in our earlier report (Mineau and Palmer 2013). The Eared Doves showed a much stronger response to imidacloprid than did Red-winged Blackbirds or pigeons tested in industry studies. Yet, this was not sufficient to prevent mortality. Despite the fact that the authors here used short presentation time and hunger to encourage consumption, one would expect food avoidance to be much less effective in the wild than in a laboratory setting because of the need for higher and faster food consumption in the face of higher

[16] This corresponds to about 0.028 mg a.i./seed and 0.012 mg a.i./seed for imidacloprid and thiamethoxam respectively. The rate for imidacloprid in sorghum reported by Addy-Orduna et al. (2022) is higher than currently labeled in the U.S.; the rate of thiamethoxam is lower than labeled (see section 1.4.2).

maintenance costs, predation, competition from conspecifics etc., all of which can alter the outcome of the bird-seed encounters. Birds showed severe signs of intoxication with all three sorghum seed treatments – e.g. "*lethargy or frenzy, uncoordinated walking, "sitting" and "fluffed up" or "lain"(sic) on the floor, motionless leaning forward, half-open wings, legs spread, and* (without) *any impulse to flee*" as reported for imidacloprid.

Interestingly, the authors found that the untreated seed they had purchased contained small amounts of imidacloprid. That untreated seeds should be contaminated by neonic residues is not unexpected given the prevalence of neonics in the agricultural seed pipelines. Other authors have similarly found contamination of organic seeds with neonics (e.g. Humann-Guilleminot et al. (2019c) in Switzerland)

The debate about the effectiveness of avoidance of treated seeds is alive and well and as new crops of naïve, unexperienced risk assessors are brought into the fray, they are prone to following ill-founded industry claims of 'repellency'. This appears to be a factor in the recent PMRA about-face on imidacloprid seed treatments (Morrissey 2022). Also, whereas it may be possible for birds to make an association between the onset of poisoning and a highly colored and visibly different food item such as a treated seed, it ignores the possibility of poisoning through contaminated food and water. In either case, to dismiss the possibility of acute poisonings ignores the inconvenient fact that we have evidence of acute poisonings under actual use conditions (section 1.3.4) and that there is evidence of impacts at the population level (section 3).

1.3.2. Treated seed are widely available to foraging birds, especially at spills.

Regulators posit that it will be difficult for birds to rapidly gather a substantial dose of seeds because they imagine birds having to work hard to seek individual seeds. It is clear that the longer it takes a bird to find and consume treated seed, the greater the chance that it will become sick and conditioned aversion will set in before it becomes incapacitated or dies. Unfortunately, this also is a naïve position to take, and it ignores published data on the wide availability of treated seed following normal agricultural practices, especially in the form of small spills, here, there and everywhere. Label warnings about cleaning up such spills are, unfortunately, not in conformity with standard agricultural practice.

Lopez-Antia et al. (2016) looked at the availability of cereal seed on the soil surface in Spain. As we reported back in 2013, seeds at sowing are widely available to birds even under conditions of 'precision sowing'.[17] Interestingly, fields that were drilled (precision sown) and then rolled had more surface seeds than fields that were simply drilled. Surface seed counts averaged 43/m^2 in the headlands (field borders) and 11/m^2 in the field center, excluding any obvious spills.[18] Spills were noted on nearby roads and in headlands; these were found to contain between 142 and 10,000 seeds/m2. This information once again suggests that the availability of seeds is unlikely to be limiting when trying to estimate ingestion rates before the onset of conditioned aversion. A total of 30 species were observed feeding on sown cereal and the seeds remained available on the fields for approximately one month after sowing.

Roy et al. (2019) added to the existing corpus of work on seed availability by looking at the presence of corn, soy and wheat in Minnesota fields, USA. They also did not use dye technology to count surface

[17] The reader is referred to our original report for a review of older studies on the subject. In the current report, we will emphasize recent work.

[18] Atkins et al. (1989) demonstrated that, without the use of fluorescent dyes, any visual count under normal daylight conditions should be considered minimum estimates and likely underestimates.

seeds so their numbers must be considered minimum estimates. Through an extensive survey, they documented visible seed spills as seen from roads running alongside farm fields. In doing so, they avoided the likely bias that is introduced when farmers are told they are being monitored. They also looked at a number of farmers to measure their ability to incorporate seed with current planting machinery (regular drills for winter cereal; precision drills for corn and soy). Some of these farmers were aware that they were being monitored; others, primarily state employees, were not. They applied treated seed to the soil surface to look at the speed of dissipation of the insecticide (clothianidin and thiamethoxam in corn; imidacloprid and clothianidin in soybean) over time and used trail cameras to monitor wildlife at simulated spills. In the two years of the study, observers detected 3-4 large spills/10,000 planted acres of corn, 15-26 spills / 10,000 planted acres of soybean and 6-7 spills /10,000 planted acres of wheat. The concentration of the active ingredients decreased by approximately two orders of magnitude in the 30 days they were left on the field – e.g. from approximately 1000 µg/g to 10 µg/g over that period.

Lennon et al. (2020a) also looked at the presence of clothianidin-treated wheat seeds on fields sown in the United Kingdom (UK) (no fluorescent dyes used here either). Seeding was carried out by the seed company or professionals. Within 24 hours post-sowing, seeds on the field surface ranged between 104.6 – 606.9 µg/g clothianidin for a target concentration of 500 µg/g. The average seed within 24 h of sowing contained 278.3 µg/g or 55.6 percent of labeled target but this may have been both as a result of loss and as a result of seeds gaining weight from ground moisture. (Reporting the information on a per seed basis would have been useful here). Under the weather conditions encountered (rain had the most effect on residue loss), clothianidin had a half-life of 4.2 days on surface seed. Emerging wheat seedlings contained on average only 5.9% of the clothianidin present on the ungerminated seed but concentrations were wildly variable ranging between 0.1 and 104.5 µg/g (mean of 4.8 µg/g). Seed clusters (small spills including many with >100 seeds) were frequent and observed on 79 percent of sampled fields.

Addy-Orduna et al. (2023) reported that for soybean seeding in Argentina[19], 9 percent of seeds were left on the soil surface. Photos were used to count seeds but without the use of fluorescent dye to ensure all seeds were counted. This amounted to 2.6 seeds/m^2 in the field centers and 3.2 seeds/m^2 in the headlands. However, at sites where the seeder was loaded, spills ranging from 2400-8800 seeds / m^2 were recorded.

In our earlier report (Mineau and Palmer 2013 section 3), we provided an extensive review of seeding methodology and bird behaviour in fields to argue that the availability of freshly seeded fields was not a limiting factor for birds. None of this has changed 10 years later. The studies reported above in the context of neonic insecticides complement decades of previous research on seed availability for birds. In reporting kills resulting from the use of imidacloprid in France, Millot et al. (2016) provided a very good review of information available for Grey Partridge (*Perdix perdix*) and Wood Pigeon (*Columba palumbus*) in seeded cereal fields especially. Where birds encountered high seed densities associated with small spills, seed intake rates dramatically increased in those species (see 1.3.3 below). We conclude as they do that the real risk to birds is not the average number of seeds present per m^2 of field as summarized in table 1.6., but rather the presence of unavoidable small spills that can result in a very quick uptake of seeds in very short order.

[19] Seeding was carried out at an agricultural research station by agricultural technicians

Table 1.6. Tabulated seed counts per m² of field in recent articles reviewed for this report. This table does not include spills.

Corn (maize) seeds/m²		Soybean seeds/m²		Cereal seeds/m²		Source and notes
Field center	Field edge	Field center	Field edge	Field center	Field edge	
0.04 (0-5)	0.10 (0-15)	0.6 (0-9)	1.5 (0-15)	7.8 (0-69)	8.4 (0-51)	Roy et al. 2019 (USA-Minnesota)
				11	43	Lopez-Antia et al. 2016 (Spain)
0.02 (0.003 – 0.08)						McGee 2018 (Industry study – Canada)
				0.9	3.7	Lennon et al. 2020a (UK)
					(8-96)	Millot et al. 2016 (France)
		2.6	3.2			Addy-Orduna et al. 2023 (Argentina) (12 and 48h post-seeding) Seeding rate of 38/m2

1.3.3. Direct evidence of treated seed consumption

Seed treatments have long been seen as a privileged way for many farmland species to obtain large doses of pesticides (Mineau and Palmer 2013). Historically, birds were poisoned by mercury-containing fungicides, organochlorines, then organophosphorus and carbamate insecticides. As mentioned above, Millot et al. (2016) reviewed published information on ingestion rates in the two species most often encountered in imidacloprid poisoning incidents in France. When seeds are available at high density such as in a spill (150-200 seeds/m²), ingestion rates were reported as 30 and 60 seeds per minute for Grey Partridges and Wood Pigeons respectively. Millot et al. (op. cit.) correctly concluded that, given high seed availability, a lethal amount of seeds could be ingested before the onset of post-ingestonal distress.

Lennon et al. (2020a) looked at the exposure of farmland birds to clothianidin-treated winter wheat seed in the UK. They placed cameras at 40 seed 'clusters' (spills of >100 seeds each) that had been created during normal seeding operations in 21 different fields. For those species known to take seeds, they found a positive relationship between bird abundance in fields and surface seed density, indicating that birds were attracted to the seed. Species showing this positive relationship included Starlings, Bunting species and, to a lesser extent, gamebirds. Fifteen species were seen ingesting treated seed. They sampled plasma from 10 species and had positive clothianidin detections in 9 of these. Four of the positive species were captured at seed clusters on camera; the other 5 were seen foraging in the fields. In one way or another, 21 different species were documented as being exposed to the seed treatments. They described two individuals of two species with high plasma levels (Yellowhammer - *Emberiza citronella* - with 69.3 µg/ml and Eurasian Tree Sparrow - *Passer montanus* - with 4.88 µg/ml) and with red dye around the bill indicative of a recent exposure showing symptoms of intoxication. However, looking at the number of seeds taken during feeding bouts recorded at seed clusters, no bird species approached either a lethal level of ingestion or even the reproductive threshold we estimated in 2013 – even assuming the species to be a sensitive one at the 5 percent tail of the relative distributions. The highest risk was shown for Wood Pigeon with individuals ingesting up to 152 seeds

in a 10 minute recording – although the average for that species was 9.4 seeds. This is considered a minimum ingestion because the individuals were not marked and could have been present for longer than the 10 min recordings. As the authors noted, the study was done in the fall when alternative food sources (e.g. weed seeds) are abundant. Spring sowing would likely present a higher risk of exposure. Fortunately, clothianidin is not nearly as acutely toxic as imidacloprid.

Having documented a large number of spills in farm fields (see Roy et al. 2019), the same research group (Roy and Coy 2020) used trail cameras to look at the attractiveness of those spills to wildlife. They looked at corn seed with two concentrations of clothianidin as well as corn with thiamethoxam and soybean treated with imidacloprid. They documented 16 species of birds and 14 species of mammals consuming seeds. An increasing number of birds and mammals consumed seed over time. Not surprisingly, birds seen consuming soybean seeds were the larger-bodied birds; i.e. Ring-necked Pheasant, Canada Goose, Turkey and American Crow. Many more species were seen eating corn. The smaller birds such as sparrows depended on larger birds cracking seeds so they could ingest fragments. Individual bouts of feeding were quite variable in terms of the number of the time spent at the spill and the number of seeds consumed, especially in the case of mid-sized to large birds. For example, looking at clothianidin-treated corn (0.50mg of clothianidin/seed), Crows ingested 1-30 seeds in bouts lasting 4-135s (N= 26 bouts on film), and Ring-necked Pheasants 1-25 seeds lasting 2-60s (N=10 bouts captured on film).

More mammals than birds were seen consuming seed. For example, deer were seen to consume up to 850 imidacloprid-treated soybean, or 650 clothianidin-treated corn in one feeding bout. It is not surprising that White-tailed Deer (*Odocoileus virginianus*) in agricultural landscapes have been reported to be highly contaminated with neonic insecticides with likely toxicological effects (Berheim et al. 2019). Birds and mammals did not eat greater quantities of seed as time passed, suggesting that the higher initial concentrations or amount of dye on the seeds were not avoided any more than they were later on. However, the authors did detect an increased feeding rate after rain. This might have been the result of reduced coloring or chemical loading but could also have indicated a preference for moistened seed. Many species of birds and mammals were seen feeding at simulated spills causing the authors to conclude: "*Our findings not only refute the idea that wild animals will not eat treated seeds, but unfortunately document that good seed stewardship practices were not always followed, despite clear warnings about dangers to wildlife on product labels.*" (Roy and Coy 2020).

In stark contrast to the above or with any prior work of this type, an industry-sponsored study (McGee et al. 2018) using similar methodology in Ontario, Canada, showed very few clothianidin-treated corn seeds (1.25 mg/seed) on the surface and even fewer birds apparently interested in the seeds. Seeds were counted by observers walking transects with no attempt to correct for visibility. These authors report an average of 0.0224 seeds/m^2 (range of 0.003 to 0.08) on average for the fields studied. No data are given on spills except that one camera was placed at a location where 36 seeds were visible. The authors reported a total of two seeds being eaten during the entire study and seen by observers – one by a crow and the other by a Blue Jay. Video cameras trained on 'spills' of 2-36 seeds in the frame showed no removal by birds. This study clearly stands out from those of independent researchers in documenting very low numbers of surface seeds and a remarkably low interest from resident birds.

Addy-Orduna et al. (2022) in a laboratory study reviewed earlier, had concluded that soybeans were unlikely to give rise to poisonings in the field. They followed up with a field study (Addy-Orduna and Mateo 2023) where both untreated and imidacloprid-treated soybean were seeded in adjoining parcels at an Argentine agricultural field station. The imidacloprid-treated seed (0.4 mg/seed) were also dyed red; the control seed were undyed but all seed, control or imidacloprid-treated had carbendazim and

thiram on them. Based on the surface seed counts 12 and 48 hours after seeding as well as seedling emergence rate, they inferred that the treated seed were being avoided by birds, primarily Eared Doves the most frequent agricultural bird species. Damage to seedlings was greater with untreated seed. Even though this effect could have been as a result of reduced insect damage, the authors believed that much of the damage to the seedlings was a result of bird activity because they were able to compare damage to untreated seed in netted vs. un-netted plots. They calculated that the concentration measured in the cotyledons (0.9 mg/kg) was lower than expected and accounted for 0.06-0.24 percent of the material applied to the seed. Heavy rain prior to germination was probably a factor. In a prior laboratory experiment, they had found a trend for a lower ingestion of cotyledons with 33 mg/kg compared to untreated seedlings but this difference was non-significant as a result of variable ingestion rates among the subjects.

It is difficult to interpret these results and assess the relevance of the results to large-bodied consumers of soybean seed such as Ring-necked Pheasant or grouse in the North American context. Eared Doves showed a great reluctance to eat any soybeans in a laboratory environment (Addy-Orduna et al. 2022). When experimental fields were seeded with soybean, bird numbers and richness declined during and immediately after sowing with greater numbers and diversity returning only at seedling emergence time. This is in stark contrast to the work of Roy et al (2019) described above where feeding visits to soybean-seeded fields not only did not decline but possibly increased after rain events. In addition, differences between seed availability in control and treated fields (used as a proxy measurement for removal by birds) at 12 hours and 48 hours post-sowing were largely the result of higher seed numbers in the control fields overt time, making interpretation of the results difficult.

1.3.4. The inconvenient record of kills in the wild.

A reported case in the UK (Barnett et al. 2002) involved the kill of 12 racing pigeons (Rock Doves) ingesting imidacloprid-treated seed.

Millot et al. (2016) recorded 101 lethal poisoning incidents between 1995 and 2014 associated with good agricultural practice and imidacloprid used as seed treatment in France. In some cases, birds were seen dropping out of the air and displaying obvious signs of neurological impairment. Most of the incidents involved Wood Pigeons and partridges but also cranes, gulls, starlings and Ring-necked Pheasants. The number of poisoning incidents associated with corn seeding dropped abruptly in 2004 following a ban on imidacloprid for that use in France. At the time, the application rate on cereal seed in France was 0.7 g of imidacloprid per kg of seed or 0.028 to 0.042 mg of imidacloprid per seed. This is relevant to current registrations in the US where up to 0.033 mg of imidacloprid is registered for an average cereal seed. We should also note that a corn seed treatment is still registered with 1.34 mg imidacloprid per corn seed.

Lovy and Pietsch (2016) reported an incident affecting several hundred Red-winged Blackbirds, and at least one Mourning Dove (*Zenaida macroura*) and one European Starling (*Sturnus vulgaris*) in rural New Jersey, USA. Flocks of blackbirds were seen feeding in nearby cornfields, and, when approached, they were slow to fly and a few were seen dropping out of the sky. Necropsies revealed that several of the dead birds had sustained impact injuries consistent with falling to the ground. Investigation revealed that the problem had not come from the cornfields but a nearby field seeded with winter wheat. Imidacloprid-treated wheat seeds were found in the gut contents. The seeds were also treated with three fungicides, all of which of low acute toxicity to birds.

Botha et al. (2018) documented the mortality of Cape Spurfowl (*Pternistis capensis*) ingesting imidacloprid-treated barley seed. Grewing Francolin (*Francolinus africanus*) were also reported to be affected. Birds were seen falling out of the sky and scrambling for cover as described by Millot et al. (2016) for Wood Pigeon incidents and Lovy and Pietsch (2016) for songbirds. Imidacloprid was detected at 7.16 µg/g in crop contents. Liver residues measured 16 ng/g and 29 ng/g in duplicate analyses.

McMillin (2017) and Rogers et al. (2019) reported approximately 26 American Goldfinches killed by imidacloprid applied as a drench treatment to trees in a residential area in California, USA. The birds had consumed seeds initially identified as grass seed but later confirmed to be elm seeds and the investigators posited that these had been contaminated following a drench application with the Malice™ formulation of imidacloprid. Clearly, the high concentration on the seeds was such that the birds died before any conditioned aversive response. Alternatively, the birds may have been unable to associate intoxication with a common food item.

It would be nice to think that we could rely on the EPA to monitor such incidents and keep abreast of the multitude of state-based reporting systems. Unfortunately, for reasons only known to that Agency, the EPA considers any incident involving fewer than 200 birds to be a 'minor' ecological incident with no detailed reporting requirements (USEPA 2022f; Attachment 2-2.). It is probably fair to say that anyone familiar with the issue of wildlife poisonings would consider this to be quite ludicrous. It is unclear from this same attachment to the EPA endangered species assessment for imidacloprid whether any such reports were obtained or taken into consideration.

Given the extent of use of imidacloprid and other neonics of lesser toxicity, the incidents reviewed here are few. However, as experience has shown, only a small proportion of all kills are ever reported, even when an efficient reporting system is in place (Mineau and Tucker 2002). Kills are most likely detected where, within a jurisdiction with a proven reporting system, a larger flocking species of bird is killed by a pesticide so rapidly that they cannot seek cover. Historically, even insecticides with a horrendous kill record- e.g. carbofuran, did not come to light through reported kills – at least not until after a number of field studies confirmed there was a problem (Mineau et al. 2012). Organophosphorus insecticides of medium toxicity – such as fenitrothion – probably had their greatest impact through delayed mortality brought about by the debilitation of exposed individuals (Busby et al. 1989). It is difficult not to see a parallel with imidacloprid. When Eared Doves were dosed by gavage, it took until their third day for their food consumption to return to normal with either imidacloprid, clothianidin or thiamethoxam (Addy Orduna 2019). Pesticide-induced anorexia was thought to be a serious problem in the case of cholinesterase-inhibiting pesticides. Small birds especially, are unable to resist toxicologically-induced fasting and this fact is also thought to be the reason for the greater sensitivity of small birds to dosing as reviewed earlier.

1.3.5. Evidence of widespread exposure of many bird species to neonics

In our 2013 report, we identified the need for a biomarker of neonicotinoid exposure in wildlife. Despite the massive sale and use of neonics worldwide, regulators have not yet made this a mandatory requirement of registration. Progress has been made, however, in detecting the presence of neonic residues in blood at low concentrations – a non-destructive measure of exposure. Researchers are also better able to detect residues in gastro-intestinal tract contents and organs in the case of kills. Taliansky-Chamudis et al. (2017) obtained limits of quantification as low as 2 ng/ml[20] but requiring as much as 500uL of blood. Hao et al. (2018) obtained detections in the pg/ml range with much lower volumes (50ul).

However, unlike cholinesterase measures common with organophosphorus or carbamate insecticides (the precursors to neonics), we still do not have a 'smoking gun' measurement to address not only of exposure but also effect. We continue to maintain that a biochemical biomarker of receptor binding needs to be developed for this class of chemistry.

It is recognized that the presence of residues in blood is short-lived. For example, Roy et al. (2019) reported that imidacloprid residues were below detection 24 hours post-dose. However, they were able to detect residues of sub lethally exposed birds for up to 15d post-exposure in muscle, liver, kidney and brain suggesting that the detection of residues in carcasses at least should be possible after a kill is reported. All in all, feces offered the best indication of recent exposure – a week at least with low exposures (1 mg/kg/day or approx. 2 percent of the putative LD_{50} per day for 7 days) and more than 2 weeks with 10.4 mg/kg/d for 7 days. Through a dosing study in Chickens, Roy et al. (2020) confirmed the usefulness of feces as a non-destructive marker of exposure.

Bean et al. (2019) carried out a similar pharmacokinetic study in Japanese Quail, dosing them with imidacloprid-treated seed. Following ingestion, they found also that imidacloprid was rapidly cleared; to below detection levels in 24 hours and with a half-life approximating 3 hours. Typically, only a small amount of ingested imidacloprid ended up in plasma (0.04-0.76 percent following a single dose of treated seed). Concentrations of the insecticide were highest in muscle and brain. However, most was excreted in the feces as metabolites and they concluded also that this is where it was easiest to confirm exposure.

Eng et al. (2021) followed the course of imidacloprid doses in Red-winged Blackbirds gavage fed with a pesticide solution. They observed a sharp, rapid increase in circulating imidacloprid concentrations, peaking at 1 hour post dose. By 24 hours post dose, birds dosed with the lower level (1 mg/kg) had imidacloprid levels back to baseline levels[21]. Much of that decline was a result of metabolism and some of the metabolites were detected for at least 48 hours in some of the birds.

Humann-Guilleminot et al. (2023) showed that clothianidin given to House Sparrows over a five-week period of exposure (0.25 mg/kg/day) was present in new feathers being grown during this period. Measured levels were highly variable (varying by approximately 10-fold from bird to bird) but did offer another minimally-invasive method of assessing exposure. It is not known how long the presence of clothianidin in feathers would last. Because entire feathers were taken including the live portion of the quills containing some blood vessels, it is not clear to us whether the insecticide was indeed 'deposited' in the feather material as postulated by the authors or merely present in the blood and live tissue within the feather. If the latter, it is possible that the level of clothianidin might be in equilibrium with plasma and, therefore, very short lived also.

As a result of these chemical advances and despite the lack of a 'smoking gun' indicative of levels causing intoxication such as a biochemical test of impairment, there is now a plethora of studies that show extensive exposure of birds in farmland.

Exposure of quail and other larger-bodied game birds

Turaga et al. (2015) provided data on bird exposure to treated seed in NW Texas, USA on Bobwhite Quail and Scaled Quail (*Callipepla squamata*), two species exhibiting declines in that region. They

[21] But not 0. Birds were found to be contaminated pre-dose which is discussed below.

trapped and euthanized individuals of both species (81 BOBW; 17 SCQU) in the late summer of 2013, the time of winter wheat sowing. They dissected the crops of the birds and looked for evidence of treated seed and analysed livers for residues. They found no evidence of any wheat seed in any of the quail crops examined. However, liver residues showed that 17 percent of trapped birds had neonic residues in their liver, primarily imidacloprid, but also clothianidin and thiamethoxam (2 and 3 birds respectively). The highest liver residue levels documented in this study (0.062 µg/g imidacloprid) were clearly much lower than reported liver levels from poisoned Grey Partridges in France (1.0 – 1.6 µg/g) (Berny et al. 1999). In the Turaga et al. study (op.cit.) there was no matching of capture location and neonicotinoid use so it is difficult to assess the significance of the results. The authors do mention that there were treated seed for sale in the study area. The authors conclude based on these results, that there is no risk of acute poisoning to quail. This claim is difficult to substantiate. Clearly, sub-lethally exposed quail are more likely to be trapped than lethally-poisoned individuals. However, they may be correct in that the amount of active ingredient per seed might be low enough to allow a mid-sized bird such as a quail to consume a sub-lethal dose and avoid a lethal intoxication.[22] The reported median lethal level for Bobwhite Quails is 152 mg/kg although serious incapacitation is possible at around 50 mg/kg (table 1.1). This represents ingestion of 820 cereal seeds to attain an LD_{50} and 270 seeds for severe intoxications. (In comparison, the corresponding levels for corn seed are roughly 21 seeds for median lethality and seven seeds for severe debilitation).

Ertl et al. (2018) reached a very different conclusion with regards to the possible involvement of neonics in quail declines in Texas. Through spatial analysis of both Bobwhite Quail numbers and neonicotinoid use, in an approach similar to that of Hallman (2014), the authors were able to separate their study period (1978-2012) into pre-neonic, early neonic, and widespread neonic periods. They were able to show that neonic use was the most important predictor of Bobwhite Quail declines. Other factors such as the extent of developed area and the extent of cultivated cropland were also associated with declines but to a lesser extent. Weather variables such as drought, temperature or precipitation either had variable or no effects. For example, precipitation showed a U-shaped response with too little or too much rainfall having a negative effect. Interestingly, neonicotinoid use had an immediate negative effect on Bobwhite Quail numbers soon after their introduction (1994-2003). The negative association between neonic use and Bobwhite Quail numbers became the norm in all regions examined in the period of more generalized use (2004-2012).

It is rather tragic that the type of analysis that led to these conclusions is no longer possible because the main uses of neonicotinoids – as seed treatments – are no longer being tracked as of 2014 (see section 4.2.3).

Bro et al. (2016) showed that the Grey Partridge was sufficiently exposed to clothianidin and thiamethoxam used as seed treatments for the compounds to be detected in their clutches. This is despite the fact that the birds came from an area dominated by cereals (winter wheat) treated primarily with pyrethroid insecticides rather than neonicotinoids.

Lopez-Antia et al. (2016) completed a detailed assessment of the exposure of treated cereal seed to Red-legged Partridges (*Alectoris rufa*) in Spain. They found that winter-sown cereal seed (barley, wheat, oat and triticale) represented more than half of the total biomass consumed by partridges daily from October to February (53-89 percent). Residues were found in a third of the bird's crops and gizzards obtained through hunting. A total of 189 birds were sampled, most coming from localities dominated by agriculture.

[22] In the U.S. at the time, we calculated based on label information that wheat seeds carried 40 times less imidacloprid per seed than corn seeds (0.033 mg/seed vs. 1.34 mg/seed).

MacDonald et al. (2018) checked for the presence of neonics in hunter-killed male turkeys in southern Ontario, Canada. They found evidence of exposure to either clothianidin or thiamethoxam (the two main neonics used as seed treatments in the region at that time) in 22.5 percent of the birds. About half of the birds sampled showed some pesticide residue, suggesting a recent association with cropland. Maximum residue levels of 0.16 ppm thiamethoxam and 0.12 ppm clothianidin were detected.

In an unpublished 2020 MSc thesis, Sundall (2020) collected livers from harvested Ring-necked Pheasant during the 2017 spring planting season in South Dakota, USA. It was estimated that 82 percent of 185 hens had consumed seed from the fields (corn, soy or wheat) although it wasn't clear whether this all from freshly planted seed or, in part from crop residue from the previous harvest. At least 13 percent of those birds showed visible dye in their crop. Clothianidin was found in all of these three seed types with corn showing the highest concentration.[23] The author concluded that the wild Ring-necked Pheasants did consume treated seed although a captive study had shown that, when given an easy choice of untreated, red-dyed or red-dyed treated seed, untreated seeds were preferred. The author also reported that the home range of the Ring-necked Pheasants he was studying was often smaller than the size of the planted area. Because Ring-necked Pheasants are highly dependent on finding agricultural seeds[24] and because of the extent of seed treatment use, he postulated that it would be very difficult for many of the birds to access untreated seeds.

Roy and Chen (2023) analysed for the presence of neonics in feces (from lekking grounds in the Spring) and livers (from hunted birds in the Fall) of Sharp-tailed Grouse (*Tympanuchus phasianellus*) and Greater Prairie-chickens (*T. cupido*) , two iconic species of North American grasslands sampled in Minnesota, USA. Neonics were detected in the majority of grouse (93 percent in feces; 90 percent in livers) and Prairie-chickens (80 percent in feces; 76 percent of livers). Imidacloprid and clothianidin were most often detected; there were a few detections of thiacloprid[25], acetamiprid and thiamethoxam. There was a positive relationship between the probability of imidacloprid presence in the fecall pellets and the proportion of land in cultivation within a 2km distance of the collection site. Imidacloprid was the dominant seed treatment used on soy and wheat. Such a relationship was not found for clothianidin which was used primarily on corn in the region. However, even birds some distance away from seeded fields showed exposure, suggesting that seeded farm fields attracted birds from long distances. Based on previous research, the authors had determined that fecal pellets could reflect exposure for at least 15 days. Additionally, some exposure was noted in early Spring before any local seeding taking place. The proportion of livers with residues in the Fall was comparable to the proportion of birds exposed in the Spring as measured by residues in fecal pellets. This was not expected because there was relatively little fall seeding for winter wheat in the area.

The authors also measured clothianidin residues ranging from 500-1620 ng/g wet weight in ingesta of Fall-harvested birds (N=3). It is not clear whether these high levels of contamination were the result of treated seed lying on the fields all summer long or a general contamination of the bird's environment and consumption of contaminated foliage and insects. The authors detected higher imidacloprid residues from a sub-population of Sharp-tailed Grouse undergoing a sharp decline but cautioned that more work was needed before inferring cause and effect.

[23] Unfortunately, an ELISA method was used for the detection of residues and there is no indication that the method was calibrated or compared among the different seed types and other matrices such as liver or spleen.

[24] The author stated that the number of seeds thought to represent a lethal level (379 seeds) was exceeded in the crops of wild birds but no details given as to what proportion of birds etc.

[25] Thiacloprid was suspended in 2014 in the U.S. Its continued presence, even if at low frequency, is interesting.

Exposure of farmland birds more generally

Humann-Guilleminot et al. (2019a) looked at the extent to which neonicotinoids had become a ubiquitous contaminant in Switzerland. They found neonics present in all conventional farm samples but, shockingly, in 93 percent of soil and agricultural crop samples from organic farms and 80 percent from ecological set-asides both of which should not have had any neonicotinoid contamination. This was despite the fact that, at the time of the study, there was a moratorium prohibiting the use of the main neonicotinoid insecticides on a number of major crops including spring cereals and any flowering crop. Also, farmers had to agree to have their fields sampled making it unlikely that this ubiquitous contamination was the result of illegal use; also, all organic farms sampled had been in organic production for at least 10 years. A high frequency of organic fields and set-asides contained two and even three different neonics, emphasizing the multiplicity of routes that could lead to contamination, whether dust, droplet or vapor drift being picked up in rain or surface runoff. Acetamiprid was encountered more frequently in organic fields and ecological set-asides than in conventional farm fields! Comparing these levels to levels found to interfere with natural insect predators, the authors concluded that this broad scale contamination could interfere with natural pest control in organic farms or ecological set-asides. Adding insult to injury, 14 out of 16 samples of organic seed obtained either from participating organic farmers or seed outlets tested positive for neonics with combined levels as high as 81.9 ppb in one sample (but with a lower median contamination level of 0.9 ppb).

House Sparrows (*Passer domesticus*) were sampled in an associated study (Humann-Guilleminot et al. 2019c). A feather was taken from each captured individual (N= 617) and three feathers from the same site were pooled for analysis. Neonicotinoids were detected in 100 percent of samples, whether from conventional or organic sites. Thiacloprid was the most frequently encountered but clothianidin was found in the highest concentrations. The former is mainly used as a foliar spray while the latter is used as a seed treatment. The authors posited that the high levels of clothianidin were detected because the deposit of the pesticide into the feathers (but see discussion above as to whether neonics are truly deposited in the feathers) occurred during autumn seeding which was allowed. Birds were frequently exposed to multiple neonics.

Eng et al. (2021) captured northbound migrating Red-winged Blackbirds (*Agelaius phoeniceus*), near Saskatoon, Saskatchewan, Canada. They detected imidacloprid in the plasma of 54 percent of the birds analysed pre-dosing (19/35). They also reported that 17 percent and 26 percent of all birds analysed (115 birds) had residues of thiamethoxam and clothianidin respectively. Although this contamination might have been picked up by the birds prior to capture, the authors posited that it might have come from a contaminated domestic water supply of contaminated seed even if from organic sources.

Graves et al. (2022) documented lower exposure levels in Tri-colored Blackbirds (*Agelaius tricolor*), a largely insectivorous species of conservation interest in California grasslands. Out of seven salvaged carcasses from colonies in proximity to silage fields on dairy farms, two (one adult and one nestling) tested positive for clothianidin. A total of 78 birds from colonies not associated with agriculture were also analysed; none showed any residues.

Exposure of birds migrating through farmland and birds at large

In order to carry out the Eng studies (2017, 2019 – see above), White-crowned Sparrows (*Zonotrichia leucophrys*) were captured at the Long Point Bird Observatory, situated on a long peninsula jutting into Lake Erie, Ontario, Canada during the northbound migration of these birds (Hao et al. 2018). Of 36 individuals, imidacloprid was detected in 28, thiamethoxam in 8, thiacloprid and acetamiprid in

4 each. Clothianidin was thought to be present in trace amounts in 6 birds but below the formally-established detection limit. It proved the most difficult compound to analyse for. One individual had residues of all 4 compounds. The maximum level of imidacloprid recorded was 0.18 ng/ml in plasma.

Humann-Guilleminot et al. (2021) measured neonics in feathers of adult and young Barn Owls (*Tyto alba*) collected in 2012 (nestlings) and 2016 (adults) in Switzerland. They detected at least one neonic in 84 percent and 88 percent of nestlings and adults respectively. Thiacloprid and acetamiprid were the most frequently detected. However, there were also detections of clothianidin and thiamethoxam. This is despite the fact that the latter two neonics had major restrictions imposed on them in 2013. In the same publication, Humann-Guilleminot et al. also documented contamination in Alpine Swift (*Tachymarptis melba*) as measured by food boluses brought to the chicks at the nest and adult plasma in 2018.[26] Again, thiacloprid and acetamiprid were the two neonics detected with thiamethoxam a distant third. All food boluses and 87 percent of adult plasma showed evidence of at least one neonic. Although levels in the boluses speak to a low level of dosing, it does reinforce the notion that neonics are now ubiquitous in the environment and that a very large number of species are exposed at some level.

Distefano et al. (2022) greatly expanded the survey for bird exposure by looking at two species feeding in aquatic foodwebs: Sandwich Terns (*Thalasseus sandvicensis*) and Mediterranean Gulls (*Ichthyaetus melanocephalus*) in Italy. Using pooled samples of feathers as previous researchers, they identified one or more neonic in all of the Full samples (N=11) and 58 percent of Sandwich Tern samples (N=36). Imidacloprid was the most frequent followed by clothianidin. Thiacloprid was the least frequently detected.

Anderson et al. (2023) looked at neonic residues in the blood of 55 species from 17 different families in different banding sites in Texas between December 2020 and February 2022[27]. A total of 294 samples were analysed. Imidacloprid was detected in 36% of samples but this may not be a true measure of occurrence because imidacloprid was the analyte with the highest detection limit. There was a higher proportion of positive exposures in Spring and Fall compared to Summer and Winter, possibly as a result of crop seeding. Positive exposure was highest in American Robins (*Turdus migratorius*) and Red-winged Blackbirds. Acetamiprid and thiacloprid were also detected in two samples each. A few birds were resampled a number of times and showed a pattern of continuous exposure. Also, given the documented short half-life of imidacloprid in plasma, results obtained in this study suggest a pattern of continuous/repeated exposure from the bird's environment. The authors did not find a link between exposure and body condition. Doing a few 'back of the envelope' calculations based on captive quail pharmacokinetics, they estimated that the most contaminated bird processed (a House Sparrow) was exposed to an estimated 2.3 – 14.5 mg/kg of imidacloprid. That dosing level is above many of the effect levels documented above.

Although impacts in a post-neonic period are expected to decline (e.g. Millot et al. 2015), it is not clear how long birds will continue to show residues. Fuentes et al. (2023) reported that residues in the blood of several bird species in SW France caught between 2020 and 2022 (a full four years since their total ban in the region) were still as high as those reported before the ban. All five neonics were detected in Common Nightingales (*Luscinia megarhynchos*), Cirl Buntings (*Emberiza cirlus*), Grey Partridges and Montagu's Harriers (*Circus pygargus*). The one surprise species was European Blackbirds (*Turdus merula*) which did not show any residues. Their earthworm diet and the long soil

[26] Nestling feathers collected between 2004 and 2017 did not show as much exposure. Only thiacloprid was detected.

[27] What a lovely occupation during a pandemic.

persistence of neonics raised the expectation of residues. Clothianidin and thiacloprid were most frequently detected in the Buntings and Nightingales suggesting remaining contamination of the farm environment or illegal use or both. In addition, Grey Partridges showed exposure to dinotefuran and nitenpyram, both of which are only supposed to be used as vet drugs on pets and not even farm animals.

Emerging new routes of exposure

Seed treatments were clearly identified in our 2013 report as key routes of exposure for birds and the information reviewed to date for this report reinforces this point. However, whereas seed treatments may provide for the most extreme and most easy to quantify exposures, there are many other routes that should be considered in carrying out a full assessment of the impacts of neonics on birds. By looking at other routes, the number of potentially exposed species increases and so does the probability that more sensitive species will be exposed.

Bishop et al. (2018) documented exposure of two hummingbird species (Rufous - *Selasphorus rufus*; and Anna's - *Calypte anna*) to imidacloprid, clothianidin and thiamethoxam used in association with blueberry fields in southern British Columbia (BC), Canada. The chemicals were detected in the cloacal fluids produced by hummingbirds feeding on nectar sources in the fields. Given the reported contamination of both pollen and nectar by neonics, perhaps that is not so surprising. Graves et al. (2019) also documented, through feather washes and carcass residues, frequent exposure of both Anna's and Black-chinned (*Archilochus alexandri*) Hummingbirds that died at rehabilitation centers. Feather washes gave more detections, with imidacloprid and acetamiprid dominating in Southern California. Bishop et al. (2020) expanded on their previous report and sampled hummingbirds from a wider area in the Fraser valley of British Columbia. Out of 49 hummingbird cloacal fluid samples in 4 separate species, 26.5% were positive for neonicotinoids, mostly imidacloprid. One sample was positive for clothianidin and acetamiprid.

It is natural to think of insectivorous birds as small passerine species and, indeed, these are typically modeled in risk assessment calculations. An interesting study of Honey buzzards (*Pernis apivorus* - Byholm et al. 2018) breeding in Finland showed how this raptor species could be exposed to neonics through the ingestion of wasp and bumble bee larvae, its preferred food source. Contamination of the insect larval source was in turn linked to the presence of oilseed rape fields within their preferred foraging radius of 2-5 km from the nest site. Thiacloprid was the main neonic identified in the blood of nestling birds, and all of the nests sampled (N=5) showed exposure.

Badry et al. (2021) documented the presence of thiacloprid in the livers of two Red Kites (*Milvus milvus*) recovered dead in Northern Germany during a study which focused primarily on anticoagulant rodenticides. The birds were found at the time of sowing in 2009 and 2015 respectively. Thiacloprid was registered in the EU until banned in 2020. It is not clear whether the exposure might have contributed to their demise but is certainly indicative of a recent exposure event. Red Kites are prone to scavenging and the authors speculate that they were likely foraging on a primary kill – either a bird or small mammal. As with organophosphorus and carbamate insecticides, even non-bioaccumulating substances can give rise to secondary poisoning if the amount contained in the prey (especially the g.i. tract content) is sufficient.

1.4. What does risk assessment say about impacts on birds

1.4. Any fairly conducted risk assessment will show that the kills that have been recorded with imidacloprid were totally expected. Calculations show that only a few seeds or a fraction of daily food intake, whether through treated seed or invertebrates in a sprayed field, can prove lethal. Another reason we should not be surprised about kills being likely is that the toxicity of first generation neonics is similar to that of several cholinesterase-inhibiting insecticides that have a track record of kills at equivalent or lower toxic potencies.

Birds are expected to experience severe sub-lethal and reproductive effects at levels that are likely, either through the ingestion of treated seeds over a short period of time or through ingestion of contaminated foods such as invertebrates. Documenting the risk of reproductive effects through ingestion of contaminated insects requires that residues be followed over time in the food source. This is beyond our scope here but it is clear that some of the effect levels obtained from the literature, especially those having to do with sperm quality will easily be breached for a long period of time by a foraging insectivore. This is the type of risk assessment we should be demanding from regulators. Where birds are obtaining residues from insects or vegetation, it becomes impossible for them to associate any post-ingestion illness with a specific food item, such as a colored seed. Insectivorous bird species are not going to stop eating insects even if the latter cause them discomfort. The very large scale of neonic use in our farm fields makes it unlikely that the birds in question can 'move next door' and avoid continued contamination by one neonic or another.

1.4.1. Chosen effect levels

As pointed out in sections 1.1 and 1.2 above, EPA's current position on endpoints to use in a risk assessment are without scientific validity. Using the lowest of a handful of LD_{50} values, or the lowest of either Bobwhite Quail or Mallard feed concentrations that gives rise to chronic toxicity does NOT offer protection to the multitude of exposed bird species. In its most recent assessment of clothianidin, for example, the EPA (USEPA 2022b; chapter 2) offers the following endpoints to test the risk to threatened and endangered species: a lethal threshold of 423 mg/kg based on the Japanese Quail; a "sub-lethal" [emphasis added] threshold of 89 mg/kg, the midpoint or MATC between an 'observed no-effect level'[28] of 63 mg/kg and a "*200% increase in mortality*" at 125 mg/kg in the House Sparrow lethality study; and a reproductive LOAEC or MATC of 329 ppm based on the Bobwhite Quail reproductive study without apparent regard to the fact that the concentration of residues in the laboratory ration of a Northern Bobwhite Quail, even if the latter was the most sensitive species in the world, has little relevance to feeding in a wild bird species with a different weight or diet. It might be laughable if the possible consequences of such bad science weren't as serious as they are.[29]

If one truly wants to fairly assess the direct risk of neonics to birds, we continue to argue[30] that the following endpoints are reasonable <u>starting</u> points based on the information reviewed in the chapters above. Indeed, based on the available body of work now available, they are likely under-protective.

[28] It is well recognized that an observed 'no-effect level' is highly dependent on sample size, let alone test conditions. An observed no-effect level of 63 mg/kg in any given acute toxicity study clearly does not mean that mortality will not occur at that dose level.

[29] Equally serious but different problems are noted with respect to aquatic invertebrates in section 2.5.1.

[30] We made similar arguments in our 2013 report and analysis of the science available at the time.

1. **The HD_5 value.** This is the most reasonable endpoint to use if we are only concerned about the mortality of exposed species. Because of the uncertainty surrounding the right scaling exponent (see section 1.1.1) we will use the unscaled value here recognizing that this will clearly under protect small sensitive species.

2. **Severe debilitation/first mortality from acute toxicity experiments.** As argued earlier, this is highly relevant to a field situation. To determine this endpoint, we used all available studies to derive a factor that can be applied to the compound-specific HD_5 values. A ratio of severe debilitation to LD_{50} is available for 18 acute studies. Inspection of these ratios suggests that there is no clear or consistent difference among compounds[31] although, realistically, samples are small for this determination. The ratios range from <0.09 to 0.64 (table 1.1). There is reason to believe that such an approach, although reasonable is still conservative and will under protect. Firstly, many of the ratios are < (less than) values because severe effects were already seen at the lowest dose given. Also, most of the studies carry out only a perfunctory observation of the dosed animals and may miss important deficits. Nevertheless, we believe that a measure of severe debilitation is a much more reasonable endpoint to use rather than the current arbitrary and unscientific use of 'lowest available LD_{50}' or sublethal thresholds determined from studies conducted on one or a few species … which seems to be the current USEPA standard. The overall average ratio of severe debilitation to LD_{50} from these 18 studies is 0.35 with 95% confidence bounds of 0.25 – 0.44. We submit that a ratio of 0.25 – or a quarter of the estimated LD_{50} for a sensitive species – is a reasonable <u>initial</u> estimate of a maximum tolerable field exposure if we are to prevent widespread debilitation or mortality in exposed species.[32] In the clothianidin example given above, EPA (USEPA 2022b) uses 89 mg/kg as a 'sub-lethal threshold'. Yet, in the same assessment, the EPA acknowledges that severe sub-lethal effects were seen in a registrant-submitted quail study at doses as low as 25 mg/kg. This included signs of toxicity such as "… *wing droop, lower limb weakness, prostrate posture, loss of righting reflex, shallow and rapid respiration, depression, reduced reaction to external stimuli…*". Clearly, the EPA does not believe that these effects are relevant for the assessment of safety to listed species. We do not believe that many avian ecologists would agree with this position.[33]

Addy-Orduna et al. (2019) while commenting on the much higher toxicity of imidacloprid compared to that of clothianidin or thiamethoxam also commented on the fact that signs of intoxication were longer lived in the less toxic compounds. The duration of an acute debilitating intoxication is obviously relevant in a risk assessment context but cannot be tackled with the data currently in hand.

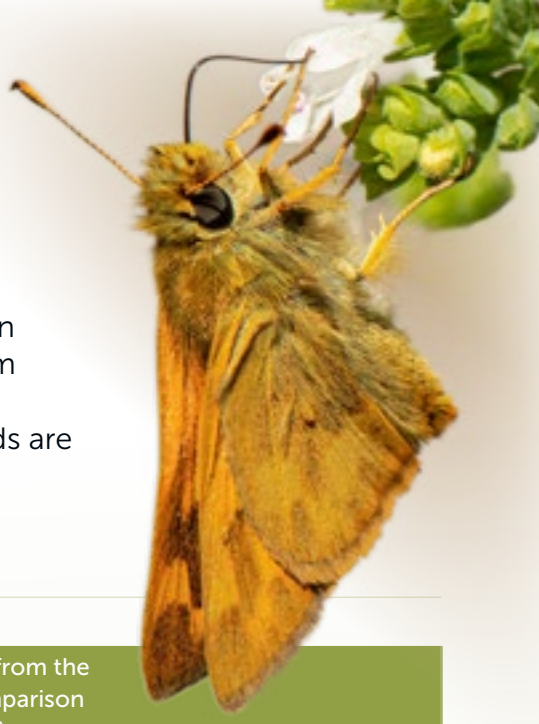
3. **The reproductive MATC extrapolated from available industry studies.**

The methodology behind this derivation was described in detail in our 2013 report: different species are expected to be as variable in their chronic toxicity threshold as they are for acute toxicity endpoints

[31] Also, the work of Addy Orduna et al. (2019) suggests there is little differentiation in the debilitation: mortality ratio among the three seed treatments they examined in the Eared Dove: imidacloprid, clothianidin and thiamethoxam.

[32] It is clear that signs of toxicity are often visible at a lower dose level that one-fourth of a lethal level. Indeed, an analysis of 164 different avian toxicity studies carried out by industry on a range of pesticides showed that 1/10 of a lethal dose would 'prevent' obvious signs of toxicity in 95% of cases (Callaghan and Mineau 2008).

[33] This also suggests that our calculated threshold 0f 29.5 mg/kg, based on ¼ of the LD_{50} will not be protective enough in all cases. In fact, the value of 25 mg/kg reported by USEPA (2022g) as causing severe debilitation in quail with clothianidin is clearly not protective enough either. With so few bird species tested, the probability that the Japanese Quail is the most sensitive bird of all is very small indeed.



(Luttik et al. 2005). We still believe that this is a more reasonable approach than assuming no species has a chronic toxicity threshold below that of the lower of Mallard or Bobwhite Quail – the EPA approach.

Table 1.7 provides our chosen acute endpoints. It is interesting to see that where we have additional information (i.e. imidacloprid), our chosen endpoint appears to be slightly under protective but not very far off from results reported by independent researchers. This does give use some confidence that chosen endpoints for the other less-studied compounds are reasonable also.

Table 1.7. Acute toxicity thresholds suitable for avian risk assessment.

Active Ingredient	Unscaled HD_5 (mg/kg)	Estimated debilitation/first mortality for sensitive species at ¼ of a lethal dose (mg/kg)	Relevant studies from the literature for comparison (see section 1.1.1.)
imidacloprid	10.2	2.6	0.93 mg/kg affected muscle strength and escape responses in Japanese Quail (Bean et al. 2019)
			1.0 mg/kg impaired hummingbird activity (English et al. 2021)
			1.2 mg/kg affected migratory ability in White-crowned Sparrows (Eng et al. 2019)
			2.5 mg/kg was the ED_{10} for severe and long-lasting neurobehavioral effects in Chickens (Franzen-Klein 2020)
acetamiprid	8.0	2.0	
dinotefuran	197	49.3	
clothianidin	118	29.5	As reported in the text above, severe life-threatening debilitation seen in Japanese Quail at 25 mg/kg or higher. The value of 29.5 mg/kg given here is offered as a reasonable way forward to fairly compare the toxicity of the various neonicotinoids given the paucity of data for some compounds.
thiamethoxam	162	40.5	
thiacloprid	467	116.8	

Chronic endpoints introduced earlier in the text are compiled in table 1.8. Again, independent studies from the literature show our chosen endpoints to be under protective for the most part.

Table 1.8. Chronic toxicity thresholds suitable for avian risk assessment

a.i.	Reproductive MATC for sensitive species (mg/kg/day) based on extrapolation from industry tests	Relevant studies from the literature for comparison (see section 1.1.1.)
acetamiprid	3.5	0.03 mg/kg (divided in 7 doses over 19 days) impaired sperm quality in House Sparrows (Humann-Guilleminot et al. 2019) Therefore average of 0.0016 mg/kg/day.
imidacloprid	2.0	0.205 mg/kg/day in Zebra Finch nestlings yields higher scaled lean body masses and higher fat levels into adulthood (Zgirski et al. 2021)
		0.77 mg/kg/day caused breeding disruption in red munia (Pandey 2017; and Pandey and Mohanti 2015, 2017)
		2.0 (1.6-2.5) mg/kg/day caused significant effects on reproduction of Grey Partridge (Lopez-Antia et al. 2015)
dinotefuran	97.0	
clothianidin	8.7	0.1 mg/kg/day caused significant increase in sperm abnormalities in Japanese Quail; decline in egg-laying starting at 1 mg/kg but statistically significant at 10 mg/kg/day (Hoshi et al. 2014)
		Effects on chick growth from pesticide deposited in the eggs seen at 42 mg/kg/day but not 8.4 mg/kg/day (Sundall 2020). This puts the MATC at 18.8.
thiamethoxam	7.3	
thiacloprid	2.6	

1.4.2. Seed ingestion as the route of exposure

We reviewed above (section 1.3) and in our 2013 report how seed treatments represent a major route of exposure to birds in farmland. This is an old problem already documented in the days of organochlorine or mercury seed treatments. Poliserpi et al. (2021b) proceeded to carry out a risk assessment for the registered rates of neonics in Argentina. Based on their assessment and given the rates of seed treatment registered in that country, most seed types were predicted to give rise to a risk of lethal intoxication, let alone sub-lethal effects. They offered several lines of evidence that a lethal dose represented a relatively small proportion of the daily intake of potentially-exposed species and that information on seed availability in fields meant that a lethal dose was available from a relatively small foraging area. The risk was highest for sorghum, corn and sunflower; somewhat less for oat and wheat and lowest for soy. Acknowledging the issue of conditioned aversion, they correctly concluded that a lethal risk of poisoning existed because birds could easily consume a lethal dose before the onset of toxicosis and development of conditioned aversion. This was essentially our conclusion based on our 2013 assessment and this position has been buttressed by numerous other risk assessments.

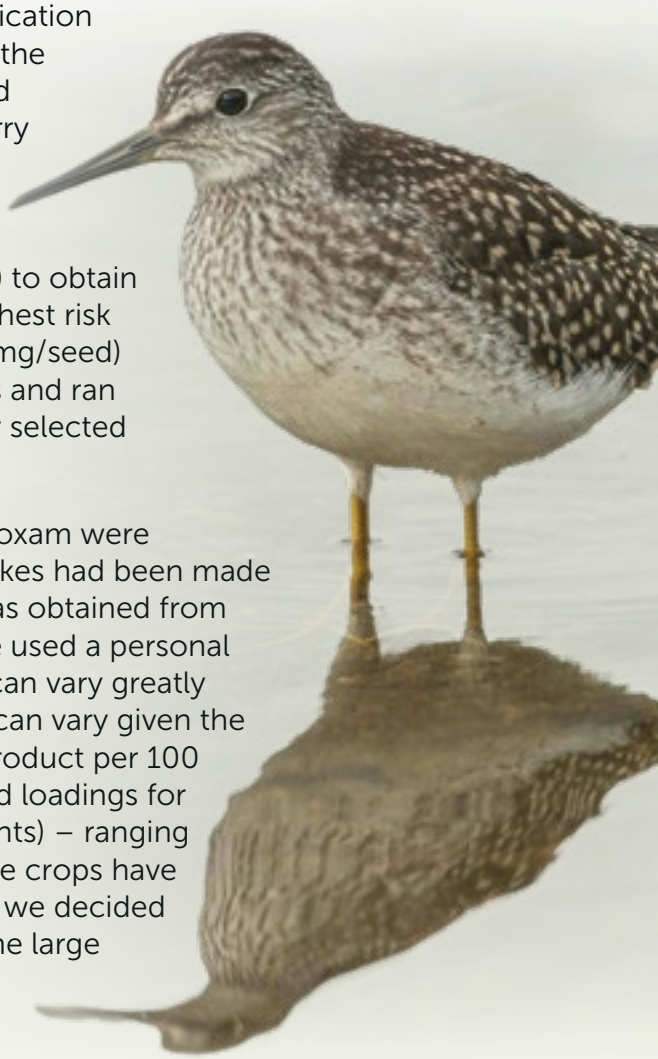
USEPA 2022 (a, b, c) appears to completely disregard the possibility that endangered species will consume treated seed. Their assessment methodology (e.g. Attachment 1-1 in all of the above references) provides information on how to estimate dose-based exposures that include dietary sources, drinking water, dermal and inhalation routes. The fact that treated seed and granular formulations have proved to be among the most important ways birds are exposed to pesticides seems to have been ignored. The standard method used by USEPA to assess the risk of granulars or seed

treatments following its usual risk assessment methods (TRex 1.15 – most recently accessed 2023) is to calculate the number of LD_{50} values per sq. ft. It is unclear whether this method was used in the endangered species assessments or whether the seed applications were merely transformed to a rate of a.i. per acre. Regardless, the LD_{50} per sq. foot concept is a highly questionable and totally unproven metric which ignores the fact that seeds are highly sought after by wildlife (section 1.3.3) and that spills are part and parcel of standard agricultural practice (section 1.3.2). Therefore, it is illusory to assume that the availability of seeds might be limiting in the case of seed-eating species. The only relevant question is how many seeds can be ingested before toxicosis sets in and feeding stops.

Nevertheless, EPA's 2022 endangered species assessment for the three main neonics (USEPA 2022f, g, h) provides a compilation of currently registered maximum rates of application to seed. However, this information appears to be at odds with industry sources. For example, the information EPA provides for clothianidin is a maximum application rate to corn seed of .000703 lb a.i./lb seed (USEPA 2022g). This works out to about 0.2 mg/seed using a corn seed of average weight. Yet, a quick review of existing industry websites – e.g. Poncho® seed treatment (https://agriculture.basf.us/crop-protection/products/seed-treatment/poncho.html - consulted 5 April 2023) describe clothianidin corn treatments ranging from 0.25 to 1.25 mg/kernel.

Similarly for thiamethoxam, the latest downloadable label for Syngenta Avicta Duo® (100-1353) recommends tank mixing with Cruiser 5FS® to achieve 1.25 mg/kernel. EPA's supposed max rate in the thiamethoxam review (USEPA 2022h) is about 0.3 mg/kernel.

For imidacloprid, EPA (USEPA 2022f) does not provide any application rates that would allow a calculated loading rate per seed. Only the rate per acre of application is provided, even in the case of seed treatments. It is difficult indeed to see how EPA manages to carry out any serious risk assessment for a seed-eating species given their apparent failure to look at the loading of the a.i. per seed. For imidacloprid, we therefore consulted the most recent labels available; e.g. USEPA Reg nos. 264-968; 228-522) to obtain information on the crops we chose to model. For corn, the highest risk identified in our 2013 report, we extracted the minimum (0.16 mg/seed) and maximum (1.34 mg/seed) from various registered products and ran both of these values through the risk assessment. For the other selected seed types and products, maximum label rates were retained.



All of the tabulated EPA use rates for clothianidin and thiamethoxam were checked against registered labels to ensure that no other mistakes had been made in the Agency compilation. For acetamiprid, the information was obtained from Reg. No. 8033-95. In order to derive loading rates per seed, we used a personal compilation of seed weights (Mineau, unpublished). Seed size can vary greatly between different varieties and therefore the loading per seed can vary given the same rate of application – often expressed as the amount of product per 100 lb. of seed. The following table provides a few examples of seed loadings for several seed types of varying size (but using average seed weights) – ranging from canola to corn and soybean (table 1.9). Typically, vegetable crops have the highest seed loadings and present a higher risk to birds but we decided nevertheless to use a few field crops as examples because of the large area over which they are used.

Table 1.9. Application rate to major seed types from USEPA 2022 b, c and selected labels assuming average seed weights.

a.i.	crop	lb a.i./lb seed (USEPA 2022 g,h & labels)	Average no. seeds / lb	mg a.i./seed given seed of average weight and maximum labeled rate
imidacloprid	field corn	-	1,681	1.34 [a]
	soy	-	2,624	0.2336 [a]
	wheat	0.0009375	13,000	0.03271
	sorghum	0.0025	72,500	0.01564
	canola	0.01	150,652	0.03011
clothianidin	field corn	0.000703	1,681	0.1898 (1.25 [a])
	rice	0.000754	21,850	0.01565
	soy	0.000502	2,624	0.08678
	wheat	0.000707	13,000	0.02467
	sorghum	0.00253	72,500	0.01583
	canola	0.00406	150,652	0.01222
thiamethoxam	field corn	0.00099	1,681	0.2672 (1.25 [a])
	rice	0.0014	21,850	0.02906
	soy	0.00075	2,624	0.1296
	wheat	0.00052	13,000	0.01814
	sorghum	0.003	72,500	0.01877
	canola	0.004	150,652	0.01204
acetamiprid	canola	0.000505313	150,652	0.001521

[a] Amount per seed already provided by the company on active labels

Exposure scenarios for seed eating birds

In order to look at the risk of ingesting treated seeds, a number of seed-eating birds were chosen to represent the wide range of bodyweights in North American species known to be attracted to agricultural seeds. These are American Goldfinch (*Spinus tristis* - 15g), Red-winged Blackbird (50g), Blue Jay (*Cyanocitta cristata* - 87g), American Crow (*Corvus brachyrhynchos* - 450g), Attwater's

Prairie-Chicken (*Tympanuchus cupido attwateri* - 737g), Ring-necked Pheasant (*Phasianus colchicus* - 1135g), Mississipi Sandhill Crane (*Grus Canadensis pulla* - 2982g) and Whooping Crane (*Grus americana* - 6400g). The Prairie-Chicken and cranes are threatened species; given that their ranges overlap potential use areas (Mineau 2014, 2015, 2018), it is useful to consider them here. To estimate the proportion of a daily intake the ingestion of seeds might represent, we used EPA methodology (USEPA 2022f, g, h – Attachment 1-1). We did not scale acute toxicity, however, given the uncertainty noted above in section 1.1.1. This will therefore tend to underestimate toxicity to small bodied birds and possibly overestimate the risk to large bodied birds. The chosen toxicity endpoints are chosen as described above.

The results for imidacloprid are presented in tables 1.10. – 1.13. The number of seeds needing to be ingested to arrive at a defined toxic threshold is given as well as the proportion of daily food intake this represents. Daily food intake was estimated by the method proposed by USEPA (T-Rex 1.5 and 2022f, g, h). Improbable scenarios are grayed out; for example, American Goldfinches are too small to be attracted to corn and soy seed; although the larger-bodied birds are unlikely to be attracted to the smallest seed types, spills might still represent feeding opportunities.

As we concluded in our 2013 analysis, the maximum rate of imidacloprid on corn is likely to have disastrous consequences to most seed-eating birds, regardless of size. Unless a great deal of the active ingredient is knocked off or washed off the seed (which can and does happen but not in a dependable way), it seems ludicrous to allow the continued use of this active ingredient on seeds. Even at the lowest possible rate registered, a lethal threshold is achieved with 1-2 seeds for a sensitive species ranging in size from a Red-winged Blackbird or Blue Jay to crows or Ring-necked Pheasants. Ingestion of less than a seed per day exceeds the reproductive MATC for a sensitive species.

Acetamiprid is only registered as a seed treatment in canola but, as expected from its high toxicity, it also has the potential to cause massive casualties. With a lethal threshold consisting of two seeds in an American Goldfinch-sized sensitive species, it is difficult indeed to see how this is deemed acceptable by regulators.

As expected, clothianidin and thiamethoxam offer less of a lethal risk. However, given the demonstrated effects on reproduction, they also can be problematic. At the current maximum label rate, less than a corn seed per day for a blackbird or jay-sized species exceeds the reproductive MATC.

If research showing effects on sperm quality holds upon further scrutiny, then a minuscule proportion of many bird's daily intake of seeds will be enough to cause effects. This is clearly worthy of further consideration.

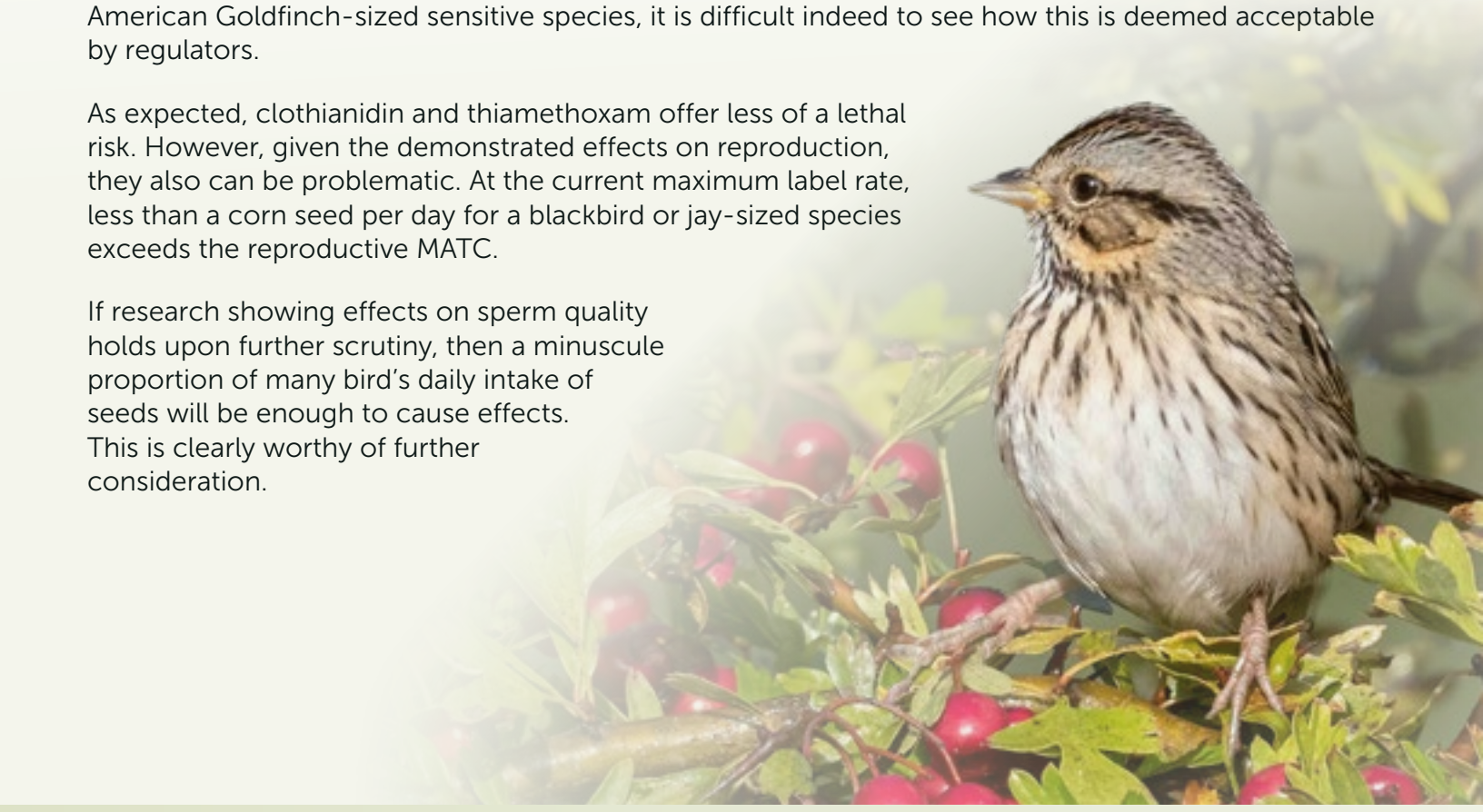


Table 1.10. Results of the risk assessment conducted for seed-eating birds of varying size ingesting imidacloprid-treated seeds of a few select crops. Improbable exposure scenarios are greyed out.

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold									
Species	Seed type	HD_5 - LD_{50} for a sensitive species (mg/kg)	Percentage of daily intake	Severe debilitation/ Lethal threshold	Percentage of daily intake	Lowest recorded effect level of note with acute dosing	Percentage of daily intake	Chronic MATC extrapolated from industry reproduction studies	Percentage of daily intake	Lowest recorded effect level of note with chronic dosing	Percentage of daily intake
		10.2 mg/kg	10.2 mg/kg	2.6 mg/kg	2.6 mg/kg	0.93 mg/kg	0.93 mg/kg	2.0 mg/ kg/ day	2.0 mg/ kg/ day	0.77 mg/ kg/day	0.77 mg/ kg/day
American Goldfinch	corn (max)	0.11	0.70%	0.03	0.18%	0.01	0.06%	0.02	0.14%	0.01	0.05%
Red-winged Blackbird	corn (max)	0.38	0.84%	0.10	0.21%	0.03	0.08%	0.07	0.17%	0.03	0.06%
Blue Jay	corn (max)	0.66	0.91%	0.17	0.23%	0.06	0.08%	0.13	0.18%	0.05	0.07%
American Crow	corn (max)	3.43	1.17%	0.87	0.30%	0.31	0.11%	0.67	0.23%	0.26	0.09%
Attwater's Prairie Chicken	corn (max)	5.61	3.20%	1.43	0.82%	0.51	0.29%	1.10	0.63%	0.42	0.24%
Ring-necked Pheasant	corn (max)	8.64	3.57%	2.20	0.91%	0.79	0.33%	1.69	0.70%	0.65	0.27%
Mississippi Sandhill Crane	corn (max)	22.70	4.54%	5.79	1.16%	2.07	0.41%	4.45	0.89%	1.71	0.34%
Whooping Crane	corn (max)	48.72	5.49%	12.42	1.40%	4.44	0.50%	9.55	1.08%	3.68	0.41%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold									
American Goldfinch	corn (min)	0.96	5.89%	0.24	1.50%	0.09	0.54%	0.19	1.15%	0.07	0.44%
Red-winged Blackbird	corn (min)	3.19	7.05%	0.81	1.80%	0.29	0.64%	0.63	1.38%	0.24	0.53%
Blue Jay	corn (min)	5.55	7.66%	1.41	1.95%	0.51	0.70%	1.09	1.50%	0.42	0.58%
American Crow	corn (min)	28.69	9.80%	7.31	2.50%	2.62	0.89%	5.63	1.92%	2.17	0.74%
Attwater's Prairie Chicken	corn (min)	46.98	26.84%	11.98	6.84%	4.28	2.45%	9.21	5.26%	3.55	2.03%
Ring-necked Pheasant	corn (min)	72.36	29.88%	18.44	7.62%	6.60	2.72%	14.19	5.86%	5.46	2.26%
Mississippi Sandhill Crane	corn (min)	190.10	38.01%	48.46	9.69%	17.33	3.47%	37.28	7.45%	14.35	2.87%
Whooping Crane	corn (min)	408.00	45.97%	104.00	11.72%	37.20	4.19%	80.00	9.01%	30.80	3.47%
American Goldfinch	soy (max)	0.65	2.58%	0.17	0.66%	0.06	0.24%	0.13	0.51%	0.05	0.19%
Red-winged Blackbird	soy (max)	2.18	3.09%	0.56	0.79%	0.20	0.28%	0.43	0.61%	0.16	0.23%
Blue Jay	soy (max)	3.80	3.36%	0.97	0.86%	0.35	0.31%	0.74	0.66%	0.29	0.25%
American Crow	soy (max)	19.65	4.30%	5.01	1.10%	1.79	0.39%	3.85	0.84%	1.48	0.32%
Attwater's Prairie Chicken	soy (max)	32.18	11.77%	8.20	3.00%	2.93	1.07%	6.31	2.31%	2.43	0.89%
Ring-necked Pheasant	soy (max)	49.56	13.11%	12.63	3.34%	4.52	1.20%	9.72	2.57%	3.74	0.99%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold									
Mississippi Sandhill Crane	soy (max)	130.21	16.67%	33.19	4.25%	11.87	1.52%	25.53	3.27%	9.83	1.26%
Whooping Crane	soy (max)	279.45	20.17%	71.23	5.14%	25.48	1.84%	54.79	3.95%	21.10	1.52%
American Goldfinch	wheat (max)	4.68	3.72%	1.19	0.95%	0.43	0.34%	0.92	0.73%	0.35	0.28%
Red-winged Blackbird	wheat (max)	15.59	4.46%	3.97	1.14%	1.42	0.41%	3.06	0.87%	1.18	0.34%
Blue Jay	wheat (max)	27.13	4.84%	6.92	1.23%	2.47	0.44%	5.32	0.95%	2.05	0.37%
American Crow	wheat (max)	140.32	6.20%	35.77	1.58%	12.79	0.57%	27.51	1.22%	10.59	0.47%
Attwater's Prairie Chicken	wheat (max)	229.81	16.97%	58.58	4.33%	20.95	1.55%	45.06	3.33%	17.35	1.28%
Ring-necked Pheasant	wheat (max)	353.92	18.90%	90.21	4.82%	32.27	1.72%	69.40	3.71%	26.72	1.43%
Mississippi Sandhill Crane	wheat (max)	929.85	24.03%	237.02	6.13%	84.78	2.19%	182.32	4.71%	70.19	1.81%
Whooping Crane	wheat (max)	1995.66	29.07%	508.70	7.41%	181.96	2.65%	391.31	5.70%	150.65	2.19%
American Goldfinch	sorghum (max)	9.78	1.40%	2.49	0.36%	0.89	0.13%	1.92	0.27%	0.74	0.11%
Red-winged Blackbird	sorghum (max)	32.61	1.67%	8.31	0.43%	2.97	0.15%	6.39	0.33%	2.46	0.13%
Blue Jay	sorghum (max)	56.74	1.82%	14.46	0.46%	5.17	0.17%	11.12	0.36%	4.28	0.14%
American Crow	sorghum (max)	293.46	2.32%	74.80	0.59%	26.76	0.21%	57.54	0.46%	22.15	0.18%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold									
Attwater's Prairie Chicken	sorghum (max)	480.62	6.36%	122.51	1.62%	43.82	0.58%	94.24	1.25%	36.28	0.48%
Ring-necked Pheasant	sorghum (max)	740.17	7.09%	188.67	1.81%	67.49	0.65%	145.13	1.39%	55.88	0.53%
Mississippi Sandhill Crane	sorghum (max)	1944.65	9.01%	495.69	2.30%	177.31	0.82%	381.30	1.77%	146.80	0.68%
Whooping Crane	sorghum (max)	4173.62	10.90%	1063.86	2.78%	380.54	0.99%	818.36	2.14%	315.07	0.82%
American Goldfinch	canola (max)	5.09	0.35%	1.30	0.09%	0.46	0.03%	1.00	0.07%	0.38	0.03%
Red-winged Blackbird	canola (max)	16.97	0.42%	4.32	0.11%	1.55	0.04%	3.33	0.08%	1.28	0.03%
Blue Jay	canola (max)	29.52	0.45%	7.52	0.12%	2.69	0.04%	5.79	0.09%	2.23	0.03%
American Crow	canola (max)	152.69	0.58%	38.92	0.15%	13.92	0.05%	29.94	0.11%	11.53	0.04%
Attwater's Prairie Chicken	canola (max)	250.07	1.59%	63.74	0.41%	22.80	0.15%	49.03	0.31%	18.88	0.12%
Ring-necked Pheasant	canola (max)	385.11	1.77%	98.17	0.45%	35.11	0.16%	75.51	0.35%	29.07	0.13%
Mississippi Sandhill Crane	canola (max)	1011.80	2.26%	257.91	0.58%	92.25	0.21%	198.39	0.44%	76.38	0.17%
Whooping Crane	canola (max)	2171.54	2.73%	553.53	0.70%	197.99	0.25%	425.79	0.54%	163.93	0.21%

Grayed Out = Scenario Unlikely

[a] See tables 1.7 and 1.8 for a review of suitable endpoints

Table 1.11. Results of the risk assessment conducted for seed-eating birds of varying size ingesting clothianidin-treated seeds of a few select crops. Improbable exposure scenarios are greyed out.

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold							
Species	Seed type	HD_5 - LD_{50} for a sensitive species (mg/kg)	Percentage of daily intake	Severe debilitation/ Lethal threshold	Percentage of daily intake	Chronic MATC extrapolated from industry reproduction studies	Percentage of daily intake	Lowest recorded effect level of note with chronic dosing	Percentage of daily intake
		118 mg/kg	118 mg/kg	29.5 mg/kg	29.5 mg/kg	8.7 mg/kg/day	8.7 mg/kg/day	0.1 mg/kg/day	0.1 mg/kg/day
American Goldfinch	corn (label)	1.416	8.716%	0.354	2.179%	0.104	0.643%	0.001	0.007%
Red-winged Blackbird	corn (label)	4.720	10.442%	1.180	2.610%	0.348	0.770%	0.004	0.009%
Blue Jay	corn (label)	8.213	11.346%	2.053	2.837%	0.606	0.837%	0.007	0.010%
American Crow	corn (label)	42.480	14.518%	10.620	3.629%	3.132	1.070%	0.036	0.012%
Attwater's Prairie Chicken	corn (label)	69.573	39.741%	17.393	9.935%	5.130	2.930%	0.059	0.034%
Ring-necked Pheasant	corn (label)	107.144	44.252%	26.786	11.063%	7.900	3.263%	0.091	0.038%
Mississippi Sandhill Crane	corn (label)	281.501	56.285%	70.375	14.071%	20.755	4.150%	0.239	0.048%
Whooping Crane	corn (label)	604.160	68.074%	151.040	17.018%	44.544	5.019%	0.512	0.058%
American Goldfinch	corn (USEPA 2021)	6.624	40.774%	1.656	10.194%	0.488	3.006%	0.006	0.035%
Red-winged Blackbird	corn (USEPA 2021)	22.080	48.845%	5.520	12.211%	1.628	3.601%	0.019	0.041%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold							
Blue Jay	corn (USEPA 2021)	38.418	53.076%	9.605	13.269%	2.833	3.913%	0.033	0.045%
American Crow	corn (USEPA 2021)	198.716	67.913%	49.679	16.978%	14.651	5.007%	0.168	0.058%
Attwater's Prairie Chicken	corn (USEPA 2021)	325.452	185.903%	81.363	46.476%	23.995	13.706%	0.276	0.158%
Ring-necked Pheasant	corn (USEPA 2021)	501.205	207.005%	125.301	51.751%	36.953	15.262%	0.425	0.175%
Mississippi Sandhill Crane	corn (USEPA 2021)	1316.824	263.293%	329.206	65.823%	97.088	19.412%	1.116	0.223%
Whooping Crane	corn (USEPA 2021)	2826.181	318.439%	706.545	79.610%	208.371	23.478%	2.395	0.270%
American Goldfinch	soy	13.654	53.829%	3.414	13.457%	1.007	3.969%	0.012	0.046%
Red-winged Blackbird	soy	45.514	64.483%	11.378	16.121%	3.356	4.754%	0.039	0.055%
Blue Jay	soy	79.194	70.069%	19.799	17.517%	5.839	5.166%	0.067	0.059%
American Crow	soy	409.625	89.656%	102.406	22.414%	30.201	6.610%	0.347	0.076%
Attwater's Prairie Chicken	soy	670.874	245.423%	167.719	61.356%	49.463	18.095%	0.569	0.208%
Ring-necked Pheasant	soy	1033.164	273.281%	258.291	68.320%	76.174	20.149%	0.876	0.232%
Mississippi Sandhill Crane	soy	2714.445	347.591%	678.611	86.898%	200.133	25.627%	2.300	0.295%
Whooping Crane	soy	5825.772	420.392%	1456.443	105.098%	429.527	30.995%	4.937	0.356%
American Goldfinch	rice	60.665	28.721%	15.166	7.180%	4.473	2.118%	0.051	0.024%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold							
Red-winged Blackbird	rice	202.216	34.406%	50.554	8.601%	14.909	2.537%	0.171	0.029%
Blue Jay	rice	351.857	37.386%	87.964	9.347%	25.942	2.756%	0.298	0.032%
American Crow	rice	1819.948	47.837%	454.987	11.959%	134.183	3.527%	1.542	0.041%
Attwater's Prairie Chicken	rice	2980.671	130.949%	745.168	32.737%	219.761	9.655%	2.526	0.111%
Ring-necked Pheasant	rice	4590.313	145.813%	1147.578	36.453%	338.438	10.751%	3.890	0.124%
Mississippi Sandhill Crane	rice	12060.189	185.461%	3015.047	46.365%	889.183	13.674%	10.220	0.157%
Whooping Crane	rice	25883.706	224.306%	6470.926	56.076%	1908.375	16.538%	21.935	0.190%
American Goldfinch	wheat	97.643	77.698%	24.411	19.424%	7.199	5.729%	0.083	0.066%
Red-winged Blackbird	wheat	325.475	93.077%	81.369	23.269%	23.997	6.862%	0.276	0.079%
Blue Jay	wheat	566.327	101.140%	141.582	25.285%	41.755	7.457%	0.480	0.086%
American Crow	wheat	2929.279	129.413%	732.320	32.353%	215.972	9.541%	2.482	0.110%
Attwater's Prairie Chicken	wheat	4797.509	354.251%	1199.377	88.563%	353.715	26.118%	4.066	0.300%
Ring-necked Pheasant	wheat	7388.294	394.462%	1847.073	98.615%	544.730	29.083%	6.261	0.334%
Mississippi Sandhill Crane	wheat	19411.359	501.722%	4852.840	125.431%	1431.176	36.991%	16.450	0.425%
Whooping Crane	wheat	41660.864	606.806%	10415.216	151.702%	3071.606	44.739%	35.306	0.514%
American Goldfinch	sorghum	94.303	13.455%	23.576	3.364%	6.953	0.992%	0.080	0.011%
Red-winged Blackbird	sorghum	314.343	16.119%	78.586	4.030%	23.176	1.188%	0.266	0.014%

		No. seeds to reach endpoint [a] and percentage of daily intake this represents for stated toxicity threshold							
Blue Jay	sorghum	546.956	17.515%	136.739	4.379%	40.326	1.291%	0.464	0.015%
American Crow	sorghum	2829.084	22.411%	707.271	5.603%	208.585	1.652%	2.398	0.019%
Attwater's Prairie Chicken	sorghum	4633.411	61.348%	1158.353	15.337%	341.616	4.523%	3.927	0.052%
Ring-necked Pheasant	sorghum	7135.579	68.312%	1783.895	17.078%	526.098	5.037%	6.047	0.058%
Mississippi Sandhill Crane	sorghum	18747.399	86.887%	4686.850	21.722%	1382.223	6.406%	15.888	0.074%
Whooping Crane	sorghum	40235.865	105.085%	10058.966	26.271%	2966.543	7.748%	34.098	0.089%
American Goldfinch	canola	145.406	9.984%	36.352	2.496%	10.721	0.736%	0.123	0.008%
Red-winged Blackbird	canola	484.687	11.961%	121.172	2.990%	35.735	0.882%	0.411	0.010%
Blue Jay	canola	843.355	12.997%	210.839	3.249%	62.180	0.958%	0.715	0.011%
American Crow	canola	4362.181	16.630%	1090.545	4.157%	321.618	1.226%	3.697	0.014%
Attwater's Prairie Chicken	canola	7144.283	45.522%	1786.071	11.381%	526.739	3.356%	6.054	0.039%
Ring-necked Pheasant	canola	11002.389	50.689%	2750.597	12.672%	811.193	3.737%	9.324	0.043%
Mississippi Sandhill Crane	canola	28906.717	64.473%	7226.679	16.118%	2131.258	4.753%	24.497	0.055%
Whooping Crane	canola	62039.903	77.976%	15509.976	19.494%	4574.128	5.749%	52.576	0.066%

Grayed Out= Scenario Unlikely

[a] From tables 1.7 & 1.8

Table 1.12. Results of the risk assessment conducted for seed-eating birds of varying size ingesting thiamethoxam-treated seeds of a few select crops. Improbable exposure scenarios are greyed out.

		No. seeds to reach endpoint and percentage of daily intake this represents for stated toxicity threshold					
Species	Seed type	HD_5 – LD_{50} for a sensitive species (mg/kg)	Percentage of daily intake	Severe debilitation/ Lethal threshold	Percentage of daily intake	Chronic MATC extrapolated from industry reproduction studies	Percentage of daily intake
		162 mg/kg	162 mg/kg	40.5 mg/kg	40.5 mg/kg	7.3 mg/kg/day	7.3 mg/kg/day
American Goldfinch	corn (label)	1.9	11.97%	0.5	2.99%	0.1	0.54%
Red-winged Blackbird	corn (label)	6.5	14.34%	1.6	3.58%	0.3	0.65%
Blue Jay	corn (label)	11.3	15.58%	2.8	3.89%	0.5	0.70%
American Crow	corn (label)	58.3	19.93%	14.6	4.98%	2.6	0.90%
Attwater's Prairie Chicken	corn (label)	95.5	54.56%	23.9	13.64%	4.3	2.46%
Ring-necked Pheasant	corn (label)	147.1	60.75%	36.8	15.19%	6.6	2.74%
Mississippi Sandhill Crane	corn (label)	386.5	77.27%	96.6	19.32%	17.4	3.48%
Whooping Crane	corn (label)	829.4	93.46%	207.4	23.36%	37.4	4.21%
American Goldfinch	corn (USEPA 2021)	9.1	55.98%	2.3	13.99%	0.4	2.52%
Red-winged Blackbird	corn (USEPA 2021)	30.3	67.06%	7.6	16.76%	1.4	3.02%
Blue Jay	corn (USEPA 2021)	52.7	72.87%	13.2	18.22%	2.4	3.28%
American Crow	corn (USEPA 2021)	272.8	93.24%	68.2	23.31%	12.3	4.20%
Attwater's Prairie Chicken	corn (USEPA 2021)	446.8	255.22%	111.7	63.81%	20.1	11.50%
Ring-necked Pheasant	corn (USEPA 2021)	688.1	284.19%	172.0	71.05%	31.0	12.81%
Mississippi Sandhill Crane	corn (USEPA 2021)	1807.8	361.47%	452.0	90.37%	81.5	16.29%

		No. seeds to reach endpoint and percentage of daily intake this represents for stated toxicity threshold					
Whooping Crane	corn (USEPA 2021)	3880.0	437.18%	970.0	109.29%	174.8	19.70%
American Goldfinch	soy	18.7	73.90%	4.7	18.48%	0.8	3.33%
Red-winged Blackbird	soy	62.5	88.53%	15.6	22.13%	2.8	3.99%
Blue Jay	soy	108.7	96.20%	27.2	24.05%	4.9	4.33%
American Crow	soy	562.4	123.09%	140.6	30.77%	25.3	5.55%
Attwater's Prairie Chicken	soy	921.0	336.94%	230.3	84.23%	41.5	15.18%
Ring-necked Pheasant	soy	1418.4	375.18%	354.6	93.80%	63.9	16.91%
Mississippi Sandhill Crane	soy	3726.6	477.20%	931.7	119.30%	167.9	21.50%
Whooping Crane	soy	7998.1	577.15%	1999.5	144.29%	360.4	26.01%
American Goldfinch	rice	83.3	39.43%	20.8	9.86%	3.8	1.78%
Red-winged Blackbird	rice	277.6	47.23%	69.4	11.81%	12.5	2.13%
Blue Jay	rice	483.1	51.33%	120.8	12.83%	21.8	2.31%
American Crow	rice	2498.6	65.67%	624.6	16.42%	112.6	2.96%
Attwater's Prairie Chicken	rice	4092.1	179.78%	1023.0	44.94%	184.4	8.10%
Ring-necked Pheasant	rice	6302.0	200.18%	1575.5	50.05%	284.0	9.02%
Mississippi Sandhill Crane	rice	16557.2	254.62%	4139.3	63.65%	746.1	11.47%
Whooping Crane	rice	35535.3	307.94%	8883.8	76.99%	1601.3	13.88%
American Goldfinch	wheat	134.1	106.67%	33.5	26.67%	6.0	4.81%
Red-winged Blackbird	wheat	446.8	127.78%	111.7	31.95%	20.1	5.76%
Blue Jay	wheat	777.5	138.85%	194.4	34.71%	35.0	6.26%
American Crow	wheat	4021.6	177.67%	1005.4	44.42%	181.2	8.01%
Attwater's Prairie Chicken	wheat	6586.4	486.34%	1646.6	121.59%	296.8	21.92%

		No. seeds to reach endpoint and percentage of daily intake this represents for stated toxicity threshold					
Ring-necked Pheasant	wheat	10143.3	541.55%	2535.8	135.39%	457.1	24.40%
Mississippi Sandhill Crane	wheat	26649.5	688.81%	6662.4	172.20%	1200.9	31.04%
Whooping Crane	wheat	57195.4	833.07%	14298.9	208.27%	2577.3	37.54%
American Goldfinch	sorghum	129.5	18.47%	32.4	4.62%	5.8	0.83%
Red-winged Blackbird	sorghum	431.6	22.13%	107.9	5.53%	19.4	1.00%
Blue Jay	sorghum	750.9	24.05%	187.7	6.01%	33.8	1.08%
American Crow	sorghum	3884.0	30.77%	971.0	7.69%	175.0	1.39%
Attwater's Prairie Chicken	sorghum	6361.1	84.22%	1590.3	21.06%	286.6	3.80%
Ring-necked Pheasant	sorghum	9796.3	93.78%	2449.1	23.45%	441.4	4.23%
Mississippi Sandhill Crane	sorghum	25738.0	119.29%	6434.5	29.82%	1159.8	5.38%
Whooping Crane	sorghum	55239.1	144.27%	13809.8	36.07%	2489.2	6.50%
American Goldfinch	canola	199.6	13.71%	49.9	3.43%	9.0	0.62%
Red-winged Blackbird	canola	665.4	16.42%	166.4	4.11%	30.0	0.74%
Blue Jay	canola	1157.8	17.84%	289.5	4.46%	52.2	0.80%
American Crow	canola	5988.8	22.83%	1497.2	5.71%	269.9	1.03%
Attwater's Prairie Chicken	canola	9808.3	62.50%	2452.1	15.62%	442.0	2.82%
Ring-necked Pheasant	canola	15105.0	69.59%	3776.2	17.40%	680.7	3.14%
Mississippi Sandhill Crane	canola	39685.5	88.51%	9921.4	22.13%	1788.3	3.99%
Whooping Crane	canola	85173.4	107.05%	21293.4	26.76%	3838.1	4.82%

Grayed Out = Scenario Unlikely

[a] From tables 1.7 & 1.8

Table 1.13. Results of the risk assessment conducted for seed-eating birds of varying size ingesting acetamiprid-treated seeds of a few select crops.

		No. seeds to reach endpoint[a] and percentage of daily intake this represents for stated toxicity threshold							
Species	Seed type	HD_5 - LD_{50} for a sensitive species (mg/kg)	Percentage of daily intake	Severe debilitation/ Lethal threshold	Percentage of daily intake	Chronic MATC extrapolated from industry reproduction studies	Percentage of daily intake	Lowest recorded effect level of note with chronic dosing	Percentage of daily intake
		8.0 mg/kg	8.0 mg/kg	2.0 mg/kg	2.0 mg/kg	3.5 mg/kg/day	3.5 mg/kg/day	0.0016 mg/kg/day	0.0016 mg/kg/day
American Goldfinch	canola	7.887	0.542%	1.972	0.135%	3.451	0.237%	0.002	0.00011%
Red-winged Blackbird	canola	26.291	0.649%	6.573	0.162%	11.502	0.284%	0.005	0.00013%
Blue Jay	canola	45.747	0.705%	11.437	0.176%	20.014	0.308%	0.009	0.00014%
American Crow	canola	236.620	0.902%	59.155	0.226%	103.521	0.395%	0.047	0.00018%
Attwater's Prairie Chicken	canola	387.531	2.469%	96.883	0.617%	169.545	1.080%	0.078	0.00049%
Ring-necked Pheasant	canola	596.809	2.750%	149.202	0.687%	261.104	1.203%	0.119	0.00055%
Mississippi Sandhill Crane	canola	1568.004	3.497%	392.001	0.874%	686.002	1.530%	0.314	0.00070%
Whooping Crane	canola	3365.266	4.230%	841.317	1.057%	1472.304	1.850%	0.673	0.00085%

[a] From tables 1.7 & 1.8

1.4.3. Exposure to spray applications

Exposure through ingestion of pesticide-contaminated insects by birds or mammals is well documented and forms a key part of risk assessment methodology. It is more uncertain than seed ingestion largely because the extent of contamination of invertebrate prey is so variable. Mineau and Callaghan (2018) provide a detailed review of how insect residues have been characterized by North American and European regulators. In the course of spray or drench applications, other foodstuffs could also become contaminated. A recent example was provided by Rogers et al. (2019) who documented a kill of American Goldfinches following a drench application of imidacloprid. They suspected that the route of exposure was through elm seeds on the ground became contaminated with the insecticide.

In order to assess the risk of exposure through consumption of contaminated insects, we used the most recent EPA guidance on the estimation of insect residues following a spray event (T-Rex 1.5). Contrary to the rather egregious way EPA derives toxicity endpoints, this appears to be a robust analysis intended to model insect residues on a treated field given the wide variation generally observed. Simulations estimate the 90th percentile of fields for either average insect residue values for that field or the higher insect values found therein (e.g. spraying hot spots – 90th percentile of insect values within a field). These values are expressed as Residues per Unit Dose (RUD) – or ppm/1 lb a.i./acre. Those values are 65 RUD and 94.4 RUD for the 90th percentile of mean and higher values respectively. The latest EPA compilation used to model risk to species at risk (2022 f, g, h) was used to derive maximum label rates for imidacloprid, clothianidin and thiamethoxam respectively. Recent representative labels were obtained for the other a.i.s. [34]

Clearly, spray applications of either imidacloprid or acetamiprid present as much of a risk to insectivorous bird species as the seed treatments present to granivorous species (table 1.14). In fact, one may argue that the risk is actually higher because birds feeding on insects [35] are much less able to form any association between being made ill and consuming their usual insect food rather than conspicuously-colored seeds. We estimate that, with imidacloprid, a small insectivore is at lethal risk by feeding on exposed invertebrates for a quarter to a third of a day; debilitation is expected in less than one tenth of a day. This is actually quite conservative because this assessment of risk is through the dietary route only. Other potential routes of exposure include drinking from contaminated puddles, inhaling pesticide fine droplets or vapour, and dermal absorption through contact with sprayed vegetation, all of which are supposed to be considered in EPA assessments (e.g. USEPA 2022 f, g, h). Also, our calculations above ignore the fact that a small-bodied insectivore is likely much more sensitive than modeled here (because toxicity was not scaled to bodyweight) and that the small songbird in question might be feeding a brood. Young chicks, especially those of altricial species are much more vulnerable to poisoning than their provisioning parents on account of their small size and high relative food intake (Mineau 2008).

Even clothianidin and dinotefuran carry some risk of incapacitation, although clearly much lower than imidacloprid and acetamiprid.

[34] We did not conduct an exhaustive survey of all registered labels; therefore, a higher labeled rate than shown here might have escaped our attention.

[35] We chose to model insectivores but it is likely that birds eating seeds or vegetation might also be at risk. It is beyond our scope to carry out a full risk assessment of all possible feeding scenarios.

Table 1.14. Summary results of a risk assessment conducted for a small (20g) insectivore feeding on insects exposed to sprays at maximum label rate. In all cases, the 90th upper percentile of surveyed fields is used – looking at the average insect concentration in that field (65 RUD) or the more contaminated insects in that field (94 RUD).

	Max. label rate (lb a.i./ acre)[a]	Daily intake (RUD of 65) (mg/kg bw) for a 20g insectivore[b]	Daily intake (RUD of 94) (mg/kg bw) for a 20g insectivore[b]	Unscaled HD_5 (mg/ kg)[c]	Unscaled debilitation threshold (mg/kg)[c]	Days of feeding in a field with RUD of 65 to reach lethal dose (days)	Days of feeding in a field with RUD of 94 to reach lethal dose (days)	Days of feeding in a field with RUD of 65 to reach debilitation (days)	Days of feeding in a field with RUD of 94 to reach debilitation (days)
imidacloprid	0.50	26.6	38.5	10.2	2.6	0.38	0.26	0.10	0.07
clothianidin	0.40	21.3	30.8	118	29.5	5.5	3.8	1.4	0.96
thiamethoxam	0.27	14.4	20.8	162	40.5	11.3	7.8	2.8	1.9
acetamiprid	0.25	13.3	19.3	8	2	0.60	0.42	0.15	0.10
dinotefuran	0.54	28.8	41.6	197	49.3	6.9	4.7	1.7	1.2

[a] Maximum one-time application rates for soil or foliar application from USEPA (2022f, g, h) for imidacloprid, clothianidin and thiamethoxam respectively; as well as USEPA Reg. No. 2749-620 (10/28/21) for acetamiprid; and USEPA Reg. No. 86203-11 (08/10/17) for dinotefuran.

[b] Based on allometric equations and insect water content used by USEPA (USEPA 2022 a,b,c)

[c] From table 1.7.

1.4.4. Imidacloprid in the context of earlier insecticides.

At this point, it is useful to do a reality check on the results presented above. Imidacloprid is not the first insecticide of moderate bird toxicity to be unleashed on the environment – although it may be the first used at this scale. Especially pertinent are the cholinesterase-inhibiting insecticides that dominated the agricultural scene for decades before the neonics supplanted them. These compounds also affect the cholinergic pathway and are well known to produce an anorexic response and conditioned aversion (Mineau 1991). Their toxicity to birds is usually well characterized, with a number of acute toxicity tests and many have been studied in a field environment. In order to keep the comparison very simple, we noted the median avian LD_{50} of different cholinesterase-inhibiting active ingredients as well as the application rate responsible for cases of avian mortality. By factoring in the application rates, we can compare the 'toxic potency' of the applications that caused problems in the field (table 1.15). It has been shown that these two parameters alone are a good indication of a pesticide's potential for causing bird mortality (Mineau 2002). Not all of these cases are strictly comparable in that they result from different types of applications and mortality of different guilds of birds. Nevertheless, it is easy to see that the predictions of field mortality from imidacloprid sprays (section 1.4.3 above) are quite reasonable given that several insecticides have caused avian mortality at much lower toxic potencies.

Table 1.15. Examples of avian field mortality seen at similar or lower 'toxic potencies' as currently allowed on imidacloprid labels for spray applications.

Active ingredient	Median avian LD_{50} (mg/kg)	Relevant label rate (g/ha)	Toxic potency: Number of median LD_{50} equivalents applied per ha	Comparative field study
imidacloprid	35.4	560[a]	15.82	Maximum label rate against which studies with other products can be compared
fenitrothion	63.4	140	2.21	Songbird mortality noted at rates of 140 g/ha or higher in forest insect control (Busby et al. 1989)
methomyl	23.7	150	6.33	Numerous cases of mortality of songbirds drinking in leaf whorls after spraying (Hommes et al. 1990)
fenitrothion	63.4	485	7.65	Bird (adult and fledgling) mortality documented in the course of locust spraying (Mullié and Keith 1993)
dimethoate	29.5	240	8.14	Numerous cases of mortality of songbirds drinking in leaf whorls after spraying (Hommes et al. 1990)
phoxim	32.2	320	9.94	Nesting failure, dead birds and decreased abundance in forest sprays (McEwen et al 1972)
chlorpyrifos	27.4	387	14.12	Several species found dead in savannah sprayed for locust control. (Mullié and Keith 1993)
acephate	146	2240	15.34	Incapacitated and dead grouse in forest applications (Zinkl et al. 1979, Richmond et al. 1979)
dimethoate	29.5	560	18.98	Mass mortality of sage grouse feeding on contaminated foliage (Blus et al. 1989)

[a] The same as 0.5 lb/acre (from table 1.11)

2. Indirect effects through terrestrial and aquatic invertebrate removal

In our 2013 report, we argued that neonic insecticides were the 'perfect storm'. Not only did they represent a direct risk to birds through lethal and sublethal intoxication, but they also affected birds indirectly through their impact on the insect food supply, particularly aquatic emergent insects. Our position has not changed and, indeed, the vast majority of new information that has emerged in the last decade supports this conclusion and increases our concern.

2.1. The critical importance of invertebrate food resources for birds

2.1. Our understanding of the importance of insects (and other invertebrates) to birds has evolved in the last decade as more attention is placed on the quality of insect food as well as the quantity. Emergent insects have been shown to be very important form a quality point of view.

Given the marked declines in both farmland and insectivorous bird species, this is a very large area of research and we cannot do it justice here. Only a few recent papers will be reviewed.

Stanton et al. (2016, 2017) studied Tree Swallows (*Tachycineta bicolor*) in intensive and less intensive agricultural areas in Saskatchewan. Tree Swallows more easily provisioned their young in grassland sites than in cropland sites as shown by higher feeding rates and more time spent in the nest corresponding to higher insect levels. Insect availability was shown to be related to biochemical health markers – antioxidant capacity and reactive oxygen metabolites – in adult and nestling plasma samples. In addition, they saw a higher return rate (indicative of a higher fledging success in the previous year) in grassland sites (41%) compared to cropland sites (25%).

Bellavance et al. (2018) presented results from a long-term study of Tree Swallows breeding in a patchwork of agricultural landscapes in southern Quebec, Canada. They looked at potential insect prey simultaneously in traps as well as in food boluses taken from chicks. They offered some interesting information on the timing, availability and abundance of prey in response to agricultural intensification and reviewed evidence to date linking bird impacts and agricultural intensification. Cultivated areas showed early seasonal abundance of some insects (e.g. pest species) but a declining availability from mid to late season. Agricultural intensification also influenced feeding choices leading to a complex situation; we alluded to this in our 2013 report a decade ago (Mineau and Palmer 2013 – section 6.1).

Michelson et al. (2018) studied Tree Swallows at both grassland and crop-dominated sites and documented the origins of their insect food using isotopic signature in blood of adults and chicks. Overall, about 75 percent of adult and nestling diets had an aquatic origin. In one year of the study, there was a larger relative contribution of aquatic insect food in association with cropland. The authors posited that grassland habitats likely had a higher proportion of suitable terrestrial sites for foraging. An alternate explanation might be a lower availability of terrestrial prey in cropland as a result of insecticide treatments. However, this differential was not seen in the second year of the study. Although both adult and chick body condition was better in grassland than in cropland sites, the good news from this research was that the importance of the wetlands in this particular study was not completely eliminated despite the likely aquatic contamination by pesticides.

A recent view of aerial insectivore declines (Spiller and Dettmers 2019) emphasized that total abundance of insects might not be the key factor driving bird declines. They reviewed recent research that suggests that quality may be more important than quantity. There is evidence that 'high quality' insects, (because of their higher omega-3 fatty acid content) are primarily of aquatic origin. As one would expect with species ranging across (and migrating through) two continents and multiple habitats, it is complicated. Although neonics feature prominently as a possible causative agent in that review, the authors conclude that evidence to date points to 'death by a thousand cuts'. Their full working hypothesis is given as follows:

"... we propose a general hypothesis that changes in availability of high-quality prey, with spatiotemporal variability across the breeding and nonbreeding grounds due to a complex interactions of multiple effects (e.g., broad application of pesticides, agricultural intensification and other land use changes, changes in climate), have variably impacted combinations of fledging success, post-fledging

survival, and nonbreeding season body condition (including associated carryover effects) of aerial insectivores, all of which contribute to spatial, temporal, and species variation in population trends. While this hypothesis lacks many details, we believe it is consistent with most of the results reviewed in this paper." (Spiller and Dettmers 2019).

Although declines in these species begun before the introduction of neonics, the fact that our 'intensity' of insect control is currently much higher (DiBartolomeis et al. 2019) needs to be considered. Also, we have grounds to suspect that aquatic insects are being broadly affected through widespread contamination of aquatic systems by neonics (see below). The impact on aquatic insects from pesticide movement from fields or direct impingement of waterbodies during application did not start with neonics. All major insecticide groups that came before neonics had the potential to negatively impact aquatic systems: namely organochlorines, then organophosphorus and carbamate insecticides, synthetic pyrethroids, and now, neonics. The difference now is that, not only is the 'insect-killing power' of individual applications increased, but through largely prophylactic uses, the scale of use is unprecedented.

2.2. More consideration of terrestrial invertebrate loss is needed

2.2. It has become increasingly clear that we are seeing a generalized decline in insects. This will have clear repercussions for consumer species such as birds. In agricultural environments, there are now a number of studies linking neonic use specifically to declines or extirpation of terrestrial insect species such as wild bee species and butterflies.

Our main emphasis in the 2013 report was on the loss of aquatic invertebrate food resources. Given what we know now, this remains an issue of primordial importance (see section 2.5). Since our last report however, there has been an intense interest and focus on the generalized decline of insect diversity and (for most species) biomass worldwide. Beforehand, these concerns were merely the subject of 'water cooler discussions' among entomologists of the type: "Why do we no longer need to clean insects from car windscreens after long car trips?" This changed abruptly in the last decade and the ongoing 'insect apocalypse' or 'insect Armageddon' has been the subject of much analysis and concern. One key element was the 2017 publication of long-term population trends from Germany indicating a 75 percent decline over 25 years in 63 small nature reserves interspersed in the German rural landscape (Hallman et al. 2017). Several other studies followed and entomologists arrived at a general consensus that, indeed, most insect populations were likely declining, probably as a result of a number of factors including intensive agricultural practices, habitat loss, and climate change to name a few hypotheses put forward.

Throughout this debate, the most robust consensus has been that insect numbers and diversity has greatly declined in agricultural landscapes. Given modern farming which, in part, relies on massive chemical inputs including the use of insecticides, it does not take long to draw a line between insect losses and the current use of neonic insecticides, currently the dominant insecticides in most of the world. Indeed, the question might be more along the lines of: "How is it that insects have managed to survive in modern agricultural landscapes (Mineau 2021)? Neonic use has resulted in a massive increase in the insect-killing potential of insecticide use patterns (Goulson et al., 2018; DiBartolomeis et al., 2019). A recent review (Gunstone et al. 2021) argued that the killing potential of pesticides to the entire soil fauna has similarly been increased if only because of the profligate use of neonic seed treatments and the fact that the vast majority of the active ingredient stays behind in the soil after the crop grows and is harvested.

As reviewed in a previous review of the peer-reviewed literature (Mineau 2020), there are already a number of field studies that have linked neonicotinoid use specifically to insect declines.[36]

For example, Gilburn et al. (2015) found a strong association between the surface area of oilseed rape (canola) treated with neonic seed treatments and significant declines in population counts for 17 species of butterflies in the UK. In a parallel to the analysis of Hallman et al. (2014) for insectivorous birds, Gilburn et al. (op. cit.) were able to show that, prior to the introduction of neonics (1985-1998), butterfly numbers were actually increasing in those same areas making it less likely that butterfly numbers simply reflected intensive cropping.

Also in the UK, Woodcock et al. (2016) found that distribution data for 62 wild bee species showed an increase in the probability of local population extinction rates in areas where neonic-treated oilseed rape had been seeded. The most affected species were those foraging in the crop.[37] This specific analysis was put forward as a model to be followed if we are to understand the consequences of agrochemical use on invertebrate communities at the landscape scale (Mancini et al. 2020). This recent methodological review argued that datasets collected by citizen scientists (such as the data from Germany that was so useful raising attention to insect declines – Hallman et al 2017), when coupled with data on insecticide use and exposure could be used to show effects on insect populations at the landscape level.[38] Unfortunately, these types of studies do not appear to hold much sway with regulators despite their paying lip service to a 'weight of evidence' approach. They may be 'mere correlations' but we believe they should carry more weight than they currently do.

Rundlöf et al. (2015) found that oilseed rape seed treatments containing both clothianidin and the pyrethroid beta cyfluthrin affected both bumble bees (*Bombus terrestris*) and the solitary bee *Osmia bicornis* in field borders of Swedish farms. Because beta-cyfluthrin is not systemic and tightly bound to soil, the authors ascribed most of the impacts they were seeing to the clothianidin component. In contrast to the above, industry-funded studies carried out in Germany (Peters et al. 2016, Sterk et al. 2016) failed to find parallel impacts from the same clothianidin & beta-cyfluthrin product. The authors commented on the fact that levels of contamination in their studies were much lower than in the Rundlöf et al. (2015). This suggests that there were more alternative resources available for the bees in the studies carried out by industry. It is probable that risk is in part determined by whether or not bees have access to ample clean floral resources away from neonic-treated fields or whether they are more restricted to feeding within the immediate field borders of treated fields.

Forister et al. (2016), working on four different long-term monitoring sites in California, linked butterfly declines to neonic use measured on a county basis. The authors were able to remove the effects of other insecticides as well to isolate the effect of neonic use; however, their pesticide data came from

[36] Pollinators and butterflies feature prominently in insect studies. They tend to be the most studied and, indeed, much of the discussion around the acceptability of neonicotinoids has centered on their impacts on pollinators, especially managed honey bees because of the direct economic consequences of hive loss.

[37] Although the details of the various pollinator assessments are beyond this report, it is useful to mention that the USEPA pollinator assessment considers risk from neonic seed treatments in canola to be low.

[38] It should be noted that this potential exists primarily for countries where there is a rich record of insect observations as well as reasonable information on pesticide use. Unfortunately, we seem to always be 'scraping the bottom of the barrel' on both counts when it comes to North American farmland. Even California, which has one of the most enviable tools for recording pesticide use, fails when it comes to insecticide use because of its failure to consider seed treatments (Mineau 2020).

the California Pesticide Use reporting system which means that seed treatments were not included in their pesticide use estimates (Mineau 2020).

Main et al. (2020) looked at native bee diversity in and around corn or soybean in rotation treated with imidacloprid or clothianidin and found a lower diversity of wild bees associated with treated fields. Even the untreated fields in their study showed the presence of neonicotinoid residues, although residue levels were lower and not as frequently detected. The presence of a diverse wildflower community proved to be the most important factor controlling wild bee abundance – but not diversity.

2.3. Increasing information on the broad contamination of the terrestrial environment surrounding fields

2.3. In our 2013 report, we commented on the ease with which neonics can move with the water – whether surface runoff or contamination of aquifers. What has also become amply clear is the extent to which terrestrial environments a long distance from treated fields can also be contaminated. The idea that pollinators can be protected by restricting the use of neonics to crops not visited by pollinators (a common strategy employed by regulators) has been shown to be highly inadequate. Field borders and other 'protected' areas that are essential for pollinators (and multitudes of beneficial insects) are typically heavily impacted when crops are treated nearby. There is currently 'no place to hide' from neonics in our farm fields.

One reason for upgrading the risk of neonicotinoids to terrestrial invertebrates is that they are much more likely to contaminate the broader landscape than previously considered. In our 2013 report, we examined the potential for these compounds to 'follow the water', that is move in the landscape through surface runoff or even through long-term contamination of groundwater. However, a number of research studies in the last decade have emphasized the fact that uncultivated areas in proximity to farm fields are heavily contaminated by neonics. This is a critical point because, following extensive (if seriously flawed – see below) risk assessments for pollinators, regulators attempted to exempt from scrutiny those crops that were not heavily visited by pollinating species. Their assessments failed to consider the movement of residues off site – whether through dust, runoff, or windblown soil – with those residues making their way into the pollen and nectar of adjacent wildflowers.

The contamination of floral resources away from cultivated fields is now well documented. For example, Stewart et al. (2014) looked at seed treatments in corn, cotton, and soybean. They found evidence of contamination of wildflowers situated 20m on average from seeded fields. Residues were detected in a quarter of wildflower samples and, when detected, averaged 10 ng/g.

Rundlöf et al. (2015) documented clothianidin in pollen and nectar of wild plants in the edges of oilseed rape fields sowed with a seed treatment.

Sánchez-Hernández et al. (2015) found several new metabolites of clothianidin and thiamethoxam in sunflower and corn nectar and/or pollen following the use of seed treatments. Because some of these metabolites are known to be of higher toxicity to insects (and vertebrates) than the parent compounds, they argued that our assessments have seriously underestimated the toxicity of pollen and nectar to non-target organisms because all degradation products have never been hitherto considered, although they are clearly present.

Botias et al. (2016) documented significant contamination of wildflowers in field edges near oilseed rape fields, with an average residue level of 14.8 ng/g of thiamethoxam, while the canola flowers in the

treated field had a much lower residue level, namely 3.26 ng/g. Higher levels still were recorded on the foliage of wild plants in the margins; again at higher levels than in crop foliage.

Long and Krupke (2016) likewise found that pollen originating from a large number of non-crop plants in field borders was contaminated with a multitude of pesticides, not all of which came from the adjoining corn and soybean fields seeded with clothianidin and thiamethoxam. Some of the pesticides were fungicides expected to act synergistically with neonicotinoid insecticides. It is important to note that pollen was collected long after sowing in order to minimize the well-documented impact of dust contamination from the seeding operation. The authors posited that the broad contamination of wild plant resources could be from wind-erodable surface soils or from surface runoff followed by uptake.

Hladik et al. (2016) also documented extensive contamination by thiamethoxam and clothianidin of native bee species taken from Colorado grasslands. They could not establish a direct correspondence between the presence of nearby crops and residues but suspected that residues had to be obtained from such crops as corn, millet, sorghum and sunflowers present within a kilometer of collection sites.

Tsvetkov and colleagues (2017) (who published their results in the prestigious journal 'Science') working near neonic-treated corn fields in Canada found that the broad contamination of wildflower pollen lasted for the entire summer period. This result was all the more remarkable for the fact that seeding operations had made use of mandated 'fluency agents' to reduce the problem of dust production. They went on to show clear impacts on worker bee survival from feeding of bee larvae at the same concentrations they were finding in the field and documented a greatly enhanced toxicity of both clothianidin and thiamethoxam in the presence of field-realistic levels of the common fungicide boscalid. This suggests that a wide range of insects – whether pollinators or pollen-eating species – in the vicinity of crop fields are likely to be similarly affected.

Finally, Bredeson and Lundgren (2019) found high residues in cover crops inter-seeded with corn; which is perhaps ironic given that the practice was intended in part to promote beneficial insect communities.

2.4. The official pollinator assessment is a fatally flawed analysis

2.4. North American regulators, namely EPA, the PMRA and California DPR have attempted to assess the risk of the three high-use neonics to pollinators. Despite considerable time and resources devoted to this assessment, the end result is scientifically highly questionable. Although reviewed more extensively elsewhere (Mineau 2020) the summary of that assessment is repeated below. One key finding is that the risk of seed treatments – typically a large proportion of total use – for pollinators at large has been completely mischaracterized by the regulators through a series of fundamental flaws in the official assessment.

The three North American regulatory bodies - the EPA (USEPA 2020a-g), California Department of Pesticide Regulation (CaDPR 2018, 2019) and Canadian Pest Management PMRA (Health Canada 2016) teamed up to produce a coordinated pollinator assessment. With their combined resources and the sheer magnitude[39] of the assessment, one might have hoped for a rigorous and scientific piece of work. Unfortunately, it is far from that. The assessment was reviewed extensively in a previously published report (Mineau 2020), especially from the point of view of seed treatments – the most

[39] The USEPA final published version of the assessment (USEPA 2020d, e, f, g, h, i, j) came in at more than 1100 pages, once the various appendices and attachments are factored in.

extensive use of neonics on this continent as well as the least well understood. The USEPA/CalEPA/PMRA assessment wrongly considered seed treatments to present minimal risks to pollinators.

As reviewed in detail, the regulatory assessments; (1) underestimated risks to wild bee species and other pollinators by relying on honey bee colony survival as a proxy for pollinator health; (2) underestimated nectar and pollen contamination levels following the use of neonic-treated seeds by assuming that the majority of crop species would have residue values at the low end of the measured spectrum; (3) ignored risks of dust from neonic treated seeds at planting, despite ample evidence that this route of exposure is highly relevant; (4) ignored exposures of bees and other pollinators to neonic-contaminated water – including, guttation fluid and puddles in or near fields sown with neonic-treated seeds – despite existing field estimates that show that these routes of exposure can completely dwarf the routes that were formally assessed; and (5) ignored risks from neonic uses on crops deemed unattractive to honey bees, despite evidence that neonic residues migrate into adjoining areas, including adjacent wildflowers that can exceed levels in the field proper (see above); (6) excluded available peer-reviewed literature from quantitative risk assessment in favor of industry studies; and (7) ignored the growing amount of field data which now links the use of neonic-treated seeds to pollinator failure on a landscape scale. The assessment completely failed, therefore, to appreciate and acknowledge the considerable and damaging effect that neonic-treated seeds are having on pollinator populations and insect prey more broadly (Mineau 2020).

Most of the honeybee hive mortality that was seen was as a result of dust production at seeding, a route of exposure that, as just mentioned, is not even considered by EPA in its assessment. Regulators like to point out that there has been some research to improve seed coatings and modify seed planters to reduce the amount of escaping dust. Unfortunately, these are left as voluntary measures and there is no evidence to date that the industry has made much progress in solving the issue (Schaafsma et al. 2019).

2.5. Indirect effects through the aquatic route

2.5. The critical situation in aquatic systems remains essentially as we described it in our 2013 report. Regulators (EPA and PMRA) continue to tinker with aquatic quality benchmarks (presumably in response to industry comments) while ignoring much of the information now clearly linking neonic contamination with widespread disruptions of aquatic ecosystems. Earlier improvements in EPA methodology now appear to have been lost as the Agency appears to have gone back on its assessment of imidacloprid in the context of ESA-listed species.

Neonics continue to be detected at concentrations that are expected to cause severe impacts in receiving waters. If a benchmark concentration is needed, then the older benchmark of 0.01 µg/L established by EPA for imidacloprid in 2016 should still be the one against which summed concentrations of neonics should be compared. There is still too little evidence that the other neonics are much safer and too many cases where multiple compounds are detected in the same samples. Increasing reports of potential synergisms between different neonics are clearly problematic as even simple additivity of effects is already proving to be a problem in the real world.

Regulators continue to consider one neonic at the time when mixtures are becoming the norm, they continue to ignore the time-cumulative effect of toxicity despite the fact that aquatic contamination is often season-long. In short, regulators should listen to the scientific community and their consensus that neonics need to be severely restricted and water contamination reduced.

As reviewed above, there is increasing evidence that aquatic habitats are critical to insectivorous birds, if not to produce the quantity of invertebrates needed for them to complete their life cycle, to provide the quality of prey needed to produce healthy broods.

2.5.1. Revisiting the issue of aquatic benchmarks

In our 2013 report, we placed a lot of emphasis on the potential for neonics to contaminate the aquatic environment as a result of their persistence and solubility characteristics. We were likely the first to point out that EPA`s methodology for assessing risks to aquatic systems was deeply flawed, starting with their setting of 'reference doses' or water concentrations indicative of a nascent impact on aquatic systems.

"*Reference doses are set using outdated methodology which has more to do with a game of chance than with a rigorous scientific process. A complete disregard for the peer-reviewed literature is a constant factor throughout the history of neonicotinoid assessments.*" (Mineau and Palmer 2013)

The complete story behind the setting of reference levels was laid out in detail in a series of reports concentrating on New York State (Mineau 2019a, b) and California (Mineau 2020). In those reports, it was argued that EPA was repeating the same errors for clothianidin and thiamethoxam (and, by extension, the other 'lesser' neonics) that it had initially committed in its assessment of imidacloprid. One key observation was that, as of 2017 and with 36 aquatic invertebrate species tested, available test data had shown a 790,000 fold difference in sensitivity to imidacloprid from the least to the most sensitive aquatic insect or crustacean. Therefore, setting any benchmark based on the "most sensitive" species for the smaller datasets with only a handful of species (i.e. all neonics other than imidacloprid) was as scientifically rigorous as a roll of the dice. In the interim, EPA did modernize its approach to make use of species sensitivity distributions in setting toxicity benchmarks but also took the opportunity to take a page from industry and to argue that the toxicity of neonics to aquatic systems was not as high as it once thought.

What is going on with USEPA assessments?

Indeed, USEPA assessments have taken a strange turn recently. Under the pretext of standardization of data (USEPA 2022 f, g, h), a number of questionable examples of data manipulation have taken place. For example, in its previous assessment of imidacloprid, the USEPA (2016) had argued (a view supported by most aquatic toxicologists) that:

" *... the effects of imidacloprid (and other neonicotinoids) on mayfly immobilization occur at substantially lower levels than lethality. Specifically, LC50 values ranged from 6.7 to 154 µg ai/L for C. dipterum and C. horaria whereas EC50 values varied from 0.77 to 32 µg ai/L for these same species.*"

And also that:

" *... immobilization is considered an ecologically relevant apical endpoint for characterizing the acute effects of pesticides, especially neurotoxic insecticides, on aquatic organisms.*" (USEPA 2016, p.74)

In its most recent assessment of the three principal neonics geared to endangered species, the EPA (2022 f, g, h) inexplicably adopted data exclusion principles of its own and, importantly, changed how study endpoints are considered. Despite the agency's previous arguments about the ecological importance of immobilization in laboratory tests, it decided (or was convinced) to favour mortality endpoints over immobilization. It also placed stricter conditions on studies, ensuring that many studies from independent university researchers would be excluded.

" ....if a definitive immobility and mortality endpoint was available from the same test, the mortality endpoint was used (because immobility is intended as a surrogate for mortality)." In addition, stricter 'quality' criteria were used such as a "...minimum of four concentrations of technical grade active ingredient, plus appropriate controls, tested within each study." (USEPA 2022f, Appendix 2-5, p.3)

Here, USEPA makes no reference to any of its previous assessments and does not discuss the reasons behind the policy changes that led to rejecting immobilization as an endpoint of concern, or the outright rejection of test data based on formulated material.[40] It could be argued that these are often more field-relevant, certainly for acute toxicity considerations. Regardless of the reason, this means that industry tests are *de facto* given more weight in toxicity assessments given that independent researchers often find it difficult to access technical grade material.

For its 2016 acute toxicity imidacloprid standard, USEPA had used immobilization values in three species of ephemeroptera which varied between 0.65 and 1.4 µg/L. The 0.65 value (from Alexander et al. 2007) was considered 'qualitative' because EPA did not have the raw data and the 0.77 value from Roessink et al. 2013 was adopted as the freshwater acute standard. These tests were done with formulated material (TEP – or typical end use products). The acute toxicity benchmark of 0.385 µg/L was obtained by applying a factor of two to this 'quantitatively acceptable' mayfly endpoint – an acknowledgment of sorts that there are undoubtedly more sensitive species that remain untested. In the same earlier assessment (USEPA 2016), a species sensitivity distribution of 32 acute values had returned an HC_5 of 0.36 µg/L. The concordance between these two values undoubtedly provided some comfort to the USEPA scientists who went on to use the 0.385 µg/L benchmark. This is the one that still appears on their website. In that same report (USEPA 2016), USEPA rejected a Bayer Crop Science attempt (cited as Moore et al. 2016) to peg an HC_5 value at 1.73 µg/L on the grounds that the company had shown clear bias in choosing which data points were acceptable.

In a spectacular reversal, and under the guise of assessing the risk to endangered species, EPA (2022f) adopted 1.43 µg/L (1.10 µg/L for insect species) and 13.15 µg/L as its acute endpoints for freshwater and saltwater invertebrate species respectively. This was based on HD_5 values after a distribution analysis of carefully-selected data (replacing immobilization by mortality when possible, eliminating tests with formulated material or too few dose levels) as well as the use of different distributions which resulted in higher HC_5 values.[41] This revisionism contrasts with the European Food Safety Authority assessment of imidacloprid which places its acute aquatic benchmark at 0.098 µg/L (EFSA 2014) or even the Canadian acute benchmark of 0.36 µg/L – also the HC_5 of available test data (PMRA 2016).[42]

[40] Industry has a near monopoly when it comes to the availability of the technical active ingredient. Using a formulated on-the-shelf product is often the only viable option available to researchers.

[41] USEPA (2021) argued that available data fit a triangular distribution rather than the more commonly used normal distribution. This has the effect of pulling in the tails of the distribution and returning higher estimated HC_5 values.

[42] Since 2001, PMRA has been concerned about the aquatic toxicity of imidacloprid and stated they were only willing to entertain new uses that were ..." *"in low environmental risk situations or critical need uses in the context of sustainable pest management programs*". Then in 2016, they proposed a complete phase out in line with the EU. It is noteworthy that the PMRA, following pressure from industry, has also increase the value of its benchmark value. In a complete reversal of the proposed cancelation of imidacloprid, the PMRA reassessed its toxicity benchmarks as 0.54 µg/L for acute effects and 0.16 µg/L for chronic. Their current approach depends entirely on mesocosm studies despite the well-known insensitivity of these systems, the underrepresentation of sensitive species and their criticism of this approach in 2016. They acknowledged a distribution-based value of 0.011 µg/L for a benchmark based on all available chronic toxicity tests but ignored this value in their risk assessment.

While USEPA (2022f) is arguing that its data selection process is meant to introduce more scientific rigour to the process of deriving benchmarks, it commits a serious methodological error by introducing multiple data points for the same species, giving them undue weight in the distribution. (viz. *"In cases where multiple endpoints were available for the same test species, the full range of applicable data is used in SSD creation."*; EPA 2022g; Appendix 2-5) For example, the highly insensitive cladoceran species *Daphnia magna* is entered six separate times in the same distribution analysis.

For its chronic toxicity standard, the USEPA (2016) had relied on data by Roessink et al. (2013) who tested the toxicity of formulated imidacloprid to two mayfly species. On that basis, they established no effect levels of 0.01 µg/L and 0.03 µg/L for the two species tested. The EPA also commented that mayflies collected in the summer appeared to be more sensitive than those collected in the winter. The chronic threshold toxicity value of 0.01 had finally brought the USEPA in line with European assessors; for example EFSA (2014) with a value of 0.009 µg/L the Dutch RIVM (2014) with 0.0083 µg/L and our earlier report (Mineau and Palmer 2013) with a value of 0.0086 µg/L obtained from the use of acute to chronic extrapolation factors. However, in its 2022 assessment of endangered species (op.cit.), the EPA rejected the aforementioned studies on mayflies (because they were performed with formulated material) and adopted benchmark of 0.125 µg/L – or 10 times less protective – from a study on a midge species. The EPA then could have performed a distribution analysis to give more credence to their chronic benchmark but, instead, reverted back to using the lowest value once they had eliminated the more sensitive results.

In our 2013 report, we argued that EPA, had seriously underestimated the aquatic toxicity of neonic residues in water and was out of step with European, and Canadian regulators as well as independent researchers. A case in point was imidacloprid. After making vast improvements from its original 1994 assessment through to 2016 (see Mineau 2019b for a detailed analysis of the evolving benchmarks), the Agency has now gone back and increased the imidacloprid benchmark 10-fold using various data manipulations detailed above. It is rather ironic that this reversal and 'downgrading' of the toxicity of imidacloprid is in the context of an assessment that is meant to protect endangered species.[43] The situation is different for clothianidin and thiamethoxam. Based on a comparison of tests carried out with the latter, USEPA (2022 g, h) appears to be using clothianidin-based benchmarks for both. The ludicrously high online thiamethoxam benchmarks were clearly the result of an inadequate dataset.

Table 2.1. Comparison of USEPA aquatic freshwater benchmarks[a] versus the latest assessment for endangered species protection (USEPA 2022f, g, h). See text for an explanation.

Active Ingredient	Acute (µg/L)		Chronic (µg/L)	
	USEPA online benchmark [a]	USEPA 2022 [b]	USEPA online benchmark [a]	USEPA 2022 [b]
imidacloprid	0.385	1.43	0.01	0.125
thiamethoxam	17.5	3.58	0.74	0.05

[43] Despite the dubious nature of the EPA assessment for reasons described above, the agency still concludes that for imidacloprid, the majority of endangered aquatic invertebrates are likely to be adversely affected based on not only the ratio of exposure to effects but also on the overlap between use areas and species ranges and habitats (USEPA2022f).

Active Ingredient	Acute (µg/L)		Chronic (µg/L)	
clothianidin	11	3.58	0.05	0.05
thiacloprid	18.9		0.97	
acetamiprid	10.5		2.1	
dinotefuran	>484150		>95300	

[a] Data obtained from: https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk (Consulted March 2023; but said to be updated to September 2022). These benchmarks are based on the USEPA 2016 analysis). However, these do not reflect more recent assessments such as the USEPA 2022 endangered species assessments.

[b] Endangered species assessments; referenced as USEPA (2022 f, g, h).

How different are the neonics from each other in their aquatic toxicity?

In our earlier report (Mineau and Palmer 2013), we had put forward the case that the aquatic toxicity of thiamethoxam and clothianidin to aquatic insects and crustacea should be assumed to be similar to that of imidacloprid based on a comparison of toxicity tests performed on the same species with different neonics. The argument was restated, and strengthened in Morrissey et al. (2015). These authors concluded that:

"*In general, acute and chronic toxicity of the neonicotinoids varies greatly among aquatic arthropods. . . . Based on limited data, however, it appears that differences in relative toxicity among the various individual neonicotinoids are minor.*" (Morrissey et al. 2015)

Other authors have also commented on the similar toxicity of the first and second generation neonics[44] (e.g., Hoyle and Code 2016) on the strength of newer data such as that of Cavallaro et al. (2017). The latter obtained comparative data for the three neonics on the same chironomid species. They found almost identical toxicities for imidacloprid and clothianidin – somewhat less for thiamethoxam.

The publication of more comparative data by Raby et al. (2018a,b) finally provided information sufficient to convince USEPA that differences between neonic active ingredients were indeed not as great as they had originally believed (USEPA 2020h).

"*When considering the toxicity data for the mayfly, all four chemicals are similar, with clothianidin, dinotefuran and thiamethoxam all having 95% confidence intervals that overlap with the confidence intervals of imidacloprid. For the midge, there are slight differences in toxicity among the chemicals, where both clothianidin and imidacloprid are similar (95% confidence bounds overlap) and dinotefuran and thiamethoxam are slightly less toxic (LC50 values are 2x and 5x higher than imidacloprid; confidence bounds do not overlap with those of imidacloprid or clothianidin).*" (USEPA 2020h)

Similar results were obtained in the chronic toxicity tests with thiamethoxam being slightly less toxic than imidacloprid – but by a two-fold difference only. It should be noted that thiamethoxam breaks down to

[44] Imidacloprid, clothianidin and thiamethoxam

clothianidin so the lesser toxicity of the former is not as relevant ecologically. No-effect concentrations for clothianidin and imidacloprid were within a factor of four and two for the most sensitive and second-most sensitive species respectively. Clothianidin was more toxic than imidacloprid to the most sensitive species (a mayfly) but less toxic than imidacloprid for the second-most toxic, a chironomid. Maloney (2018b) found that under simulated field conditions, chironomid populations were equally affected by imidacloprid and clothianidin while thiamethoxam appeared to be about 10 times less toxic.

Clearly, the differential toxicity ascribed to the three main neonic active ingredients in past and present EPA aquatic risk assessments is not justified scientifically. The current differential between imidacloprid and the second generation clothianidin and thiamethoxam operating in opposite directions for acute and chronic values (see table 2.1 above) points to a rather implausible situation, and reflects the ongoing inadequacy of EPA methodology. We posit that, when it comes to protecting aquatic species assemblages, neonics should be considered to be of equivalent toxicity until proven otherwise. This includes dinotefuran, the most data-poor active ingredient. Given that water samples typically contain several neonic residues, an additive model of effect is as good a place as any to start evaluating the real impacts of neonics.

In the previous reviews and analyses referenced above (Mineau and Palmer 2013, Morissey et al 2015, Mineau 2019 a, b, Mineau 2020) we argued that, because of their persistence (demonstration of season-long presence in monitored bodies of water) and near-cumulative effects shown in invertebrate tests, the chronic benchmark is the ecologically-relevant one to use when assessing risk from monitored water concentrations. We stand by that assessment.

'Structural' issues still plague the EPA assessment of neonics in aquatic systems

Aside from the serious issues discussed above, there are still fundamental problems in the way neonics are being considered by the EPA in aquatic risk assessments. These fundamental issues were discussed in our 2013 report; they have yet to be addressed or even acknowledged by EPA.

The most important issue is the ongoing failure to address the time dependence of neonic toxicity. As reviewed in our earlier report, Tennekes (2010) was the first to argue that neonics behave as 'one-hit' chemicals – that is, they show almost perfect cumulative toxicity. This means that a small dose can be as hazardous as a larger one provided exposure time is extended. This argument has been restated several times, lastly by Sánchez-Bayo and Tennekes (2020). Neonic residues are detected in watersheds for more than a year after application. Even chronic toxicity benchmarks (which are based on 21-28 day tests) are therefore under-protective. Following this logic, impacts to aquatic life are expected at levels far below chronic toxicity thresholds. Also, as reviewed in our earlier report, there is experimental evidence that even short pulses of neonics give rise to delayed mortality in exposed aquatic invertebrates. This delayed mortality is not captured in the current test protocols.

Both of these issues present a serious challenge to the way neonics are currently being assessed. Yet, to our knowledge, EPA and other regulators carry on as if these facts did not exist. They have had more than a decade now to address these issues. Why is this not happening?

Secondly, EPA persists in assessing the toxicity of neonics to freshwater and saltwater organisms separately. We made the point in 2013 that available science did not support this position. The lower apparent sensitivity of saltwater or brackish species is an artifact of a dearth of toxicity data. We believe that this places species-rich estuaries and other coastal areas at much greater risk than is currently acknowledged.

2.5.2. How is the aquatic risk of neonics currently viewed in the wider scientific community?

There has been so much new information and analysis since our 2013 report that a complete review of exposure and effects in aquatic systems is beyond the scope of this report. Rather than an exhaustive primary review of the aquatic literature, emphasis here will be on citing key review and assessment documents.

One notable analysis which came close on the heels of our earlier report (Mineau and Callaghan 2013) was the Worldwide Integrated Assessment of the Impact of Systemic Pesticides on Biodiversity and Ecosystems (WIA). This group of scientists from a large number of countries and institutions reviewed the very large body of science on neonicotinoid insecticides[45] that was available at the time. In their review of aquatic ecotoxicology (Pisa et al. 2014) they concluded that, at realistic levels of water contamination, it would be reasonable to expect deleterious effects on the physiology and survival of a wide range of species, whether in terrestrial, fresh-water or marine habitats. Chagnon et al. (2015) continued the same analysis and concluded that the declines of emergent invertebrate prey resulting from insecticide use was a plausible cause of some population declines of insectivorous bird species.

Morrissey et al. (2015) published the first broad-scale quantitative risk analysis by comparing literature-based effect benchmarks to the growing amount of information on residue levels in waterbodies. They concluded that 81 percent and 74 percent of maximum and average individual neonicotinoid concentrations exceeded the benchmarks that were developed, namely 0.2 µg/L (acute) and 0.035 µg/L (chronic) in their estimation. They stressed that the situation is actually worse than this because several neonicotinoids are often detected together so that one should really compare effect benchmarks to the summed concentration of neonics. They concluded that both short-term and long-term impacts of neonics were occurring at very broad geographical scales.

Sanchez-Bayo et al. (2016) offer a similar conclusion:

"*Negative impacts of neonicotinoids in aquatic environments are a reality.... The decline of many populations of invertebrates, due mostly to the widespread presence of waterborne residues and the extreme chronic toxicity of neonicotinoids, is affecting the structure and function of aquatic ecosystems. Consequently, vertebrates that depend on insects and other aquatic invertebrates as their sole or main food resource are being affected.*" (Sanchez-Bayo et al. 2016)

The most recent attempt at a global analysis appears to be that of Wang et al. 2022. As with some of the examples above, they derived both acute and chronic benchmarks by generating species sensitivity distributions. One major difference in their approach is that they combined toxicity data from all available aquatic taxa (namely: algae, amphibians, crustaceans, fish, insects, molluscs and worms) to derive freshwater ecosystem HC_5 values. Plotted values therefore ranged over about six orders of magnitude. When the chronic data were inadequate to support a distribution, acute-chronic ratios were used to derive chronic toxicity data from acute data. This is an approach we also employed in our earlier report (Mineau and Palmer 2013). One advantage of including all taxa in a species distribution is that more data are available to plot. However, a clear disadvantage is that the mechanism of toxicity will differ among the different groups on the distribution, and we do not believe that it is appropriate to include them all on the same plot. Nevertheless, their results are presented in table 2.2 below. Clearly, their ecosystem-wide derivation of HC_5 values would seriously under protect sensitive groups such as crustaceans and insects. Possibly for that reason, the authors advocate applying a safety factor

[45] The insecticide fipronil was also included in the assessment

of 5 to derive their benchmarks from the sensitivity distributions. Because it is often recognized that HC_5 values will not in fact be conservative enough[46], many jurisdictions apply such safety factors; e.g. European regulators.

Table 2.2. Ecosystem-wide derived HC5 values and proposed benchmarks by Wang et al. 2022.

Compound	Acute HC_5 (µg/L)	Chronic HC_5 (µg/L)	Proposed acute benchmark (µg/L)	Proposed chronic benchmark (µg/L)
Acetamiprid	3.31	NA	0.662	0.0062
Clothianidin	8.94	0.039	1.79	0.0077
Dinotefuran	23.4	NA	4.67	0.0164
Imidacloprid	2.71	0.030	0.540	0.0059
Thiacloprid	3.01	0.003	0.601	0.0006
Thiamethoxam	23	0.078	4.59	0.0156

Although we do not support the principle of a single all-encompassing toxicity distribution as performed by Wang et al. (op. cit.), one interesting take-home message from this benchmark derivation is the similarity in the chronic benchmark (the ecologically-relevant one as argued earlier) among all but one of the compounds (all within a factor of three). The proposed value of 0.0059 µg/L for imidacloprid is close to but slightly more conservative than existing European benchmarks (viz. the latest EU benchmark of 0.009 µg/L – EFSA 2014). The compound which stands alone is thiacloprid, rated as being much more toxic than the others. When the authors compared their proposed benchmarks to measured water concentrations reported from the world-wide literature, they found that there were no acute risks (not surprising since their derivation method clearly under-protects) but that chronic risks were often exceeded with thiacloprid and acetamiprid predicted to have the greatest impact followed by imidacloprid, clothianidin and thiamethoxam. Only dinotefuran was predicted to present a 'moderate' risk to aquatic ecosystems.

2.5.3. Continued evidence of widespread contamination

The EPA in its 2016 review of imidacloprid had already concluded that imidacloprid levels are frequently above levels at which aquatic invertebrate species are negatively affected. They added that several key taxonomic groups of aquatic invertebrates, not merely the most sensitive ones, were likely to be adversely affected given current measured concentrations – and that is before we consider the frequent presence of other neonics in the same samples.

"... the risk findings for freshwater aquatic invertebrates do not depend solely on the high acute and chronic sensitivity of mayflies to imidacloprid. Rather, acute and chronic EECs exceed toxicity values for species distributed among multiple taxonomic groups of aquatic invertebrates." (USEPA 2016)

This conclusion was based on both effect levels and predicted exposures – the two components of a risk assessment. EPA scientists were encouraged by the fact that actual water measurements fit their

[46] Reasons for this are that, for chronic effects especially, test endpoints in often in the form of No Adverse Effect Levels. These are statistically weak constructs highly dependent on test design.

modeled levels very well. They estimated then that 60 percent of seed treatment applications, 90 percent of soil applications, and 100 percent of foliar applications of imidacloprid were expected to produce surface water contamination levels above the 0.01 µg/L benchmark. As reviewed above, all of this was rejected in their latest endangered species assessment (USEPA 2022f).

Morrissey et al. (2015) summarized information published world-wide to show that aquatic contamination of aquatic systems is unavoidable given the current use patterns and the sheer quantity of neonics in use. The following is not a comprehensive review of studies that have appeared since then but merely a few highlights.

Clearly, contamination of wetlands is fully expected and can be 'excused' when applications are directly into the wetland proper or onto seasonally-drained areas (e.g. Evelsizer and Skopec 2018 in field crops in Iowa; Hayasaka et al. 2019 in Japanese rice paddies). Samson-Robert et al. (2014) found levels as high as 55.7 µg/L clothianidin and 63.4 *µg/L* thiamethoxam in puddles on seeded fields. The risks to any aquatic organisms inhabiting these seasonal or ephemeral wetlands is clear; in addition, these very high levels raise the obvious possibility of water intake as a significant source of exposure for both vertebrate and invertebrate wildlife.

The problem (of one of the many problems) with neonics is that contamination extends far and wide as a result of the compounds' persistence and solubility characteristics. For example, Anderson (2013) found levels as high as 225 µg/L of thiamethoxam in the playa lakes of North Texas. Main et al. (2014) reported clothianidin values as high as 3.1 µg/L and thiamethoxam values as high as 1.49 µg/L from small wetlands following the use of canola seed treatments nearby.

Schaafsma et al. (2015) measured levels as high as 16.2 µg/L clothianidin and 7.5 µg/L thiamethoxam in ditches outside a corn-seeded field and 3.25 µg/L clothianidin and 16.5 µg/L thiamethoxam in puddles as far as 100 m from the fields. In a subsequent study (Schaafsma et al. 2019) they observed maximum concentrations of 6.95 *µg/L* clothianidin and 2.63 *µg/L* thiamethoxam in tile drain water. Median concentrations of clothianidin were still 0.35 *µg/L* in tile drainwater and 0.68 *µg/L* in ditches receiving tile drain inputs. This was from field applications estimated to be a mere 19 g/ha.

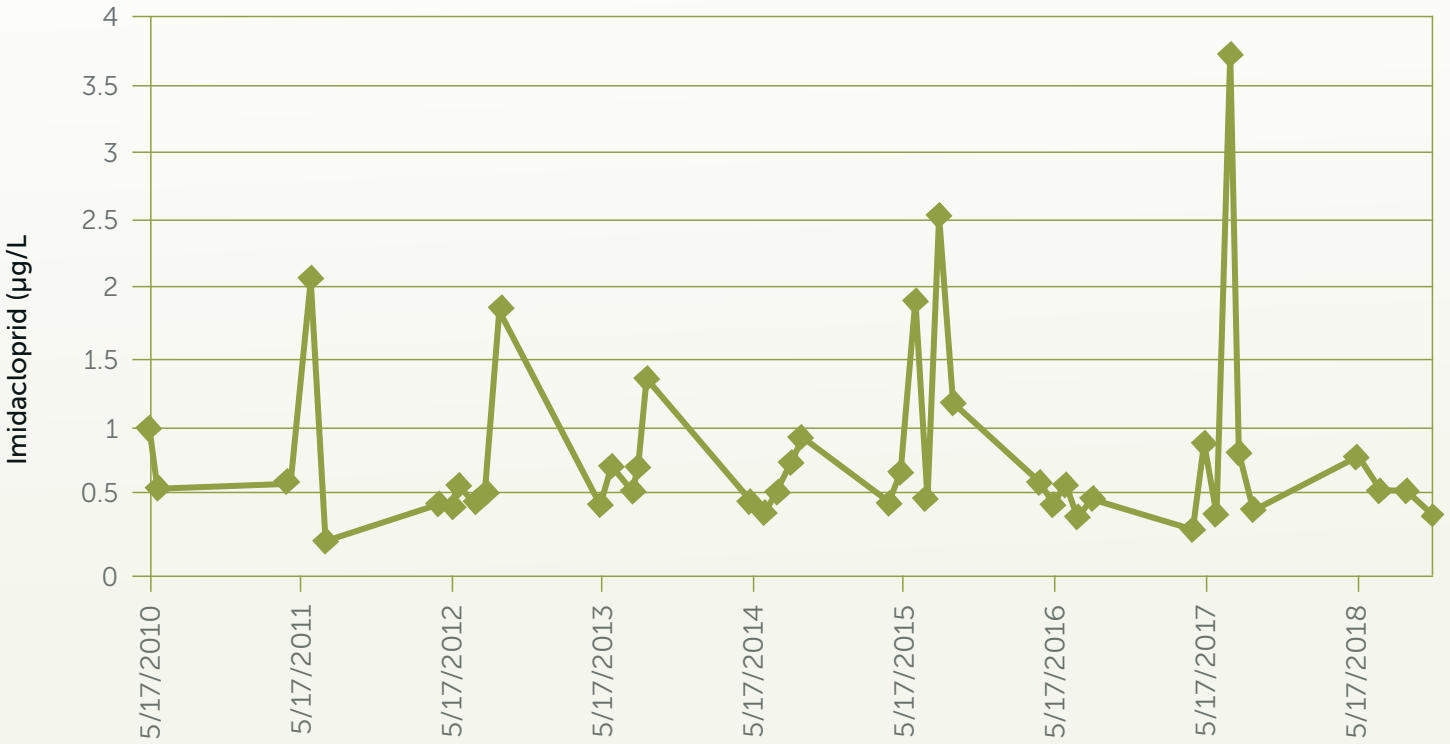
Samples taken from a variety of waterbodies in crop and non-crop sites within an agricultural landscape in Indiana (Miles et al. 2017; with 2018 correction) detected concentrations of clothianidin as high as 0.45-0.67 µg/L in small lentic woodland bodies of water well away from the seeded corn and soybean fields being monitored. Interestingly, levels in these wetlands were actually higher than those reported in any of the ditch samples taken nearer the seeded fields.

Cavallaro et al. (2019) reported values as high as 0.035 *µg/L* clothianidin and 0.23 *µg/L* thiamethoxam from wetlands within the canola-growing area of Saskatchewan, Canada.

In several studies, levels of contamination far above benchmark levels were reported early in the season, before any neonic use (seeding usually) had taken place. Occasionally, the highest reported levels are actually measured pre-seeding (e.g. Schaafsma et al. 2015) indicating a season-long level of contamination. As discussed above, it is clear that extending the exposure period dramatically increases the risk of adverse effects because a clear relationship between toxicity and duration of exposure has been shown for several neonics; i.e., more toxicity expressed with increasing length of exposure. At the risk of repeating ourselves: This has not been factored into current assessments; chronic ecological impact studies are carried out over the course of a few weeks only, while field data show that common exposure periods for wildlife are months to years, preventing any potential recovery of affected systems. In addition, sub-lethal effects such as feeding disruption, behavioral effects, and delayed development have also not been fully factored into the ecological effects of neonics.

In previous reports, one of us (PM) analysed water monitoring data for New York State (Mineau 2019 a, b) and California (Mineau 2020) only to conclude that exceedances of aquatic toxicity benchmarks were frequent. Others have arrived at the same conclusion (e.g. Hoyle and Code 2016). However, one of the most critical pieces of information is missing from many such analyses that merely report the proportion of samples that exceed safe concentrations: the repeated nature of these exceedances at many sampling sites. This is critical to our understanding of the full impacts of neonics. We made this point in our 2013 report by rearranging the data of Starner and Goh (2012) from several California watersheds. Another example, also from California was written up in Mineau (2020). It is reproduced below in figure 2.1. It shows that, over the eight years that imidacloprid was measured in Quail Creek, measured concentrations between May and November seldom dipped below 0.5 µg/L – 50 times over a 0.01 µg/L benchmark. Should it therefore be surprising that there is an increasing frequency of reports linking neonics to actual field impacts.

Figure 2.1. Imidacloprid residues at site 27-7 (Quail creek), a tributary to the Salinas River, Monterey County.



It is useful to point out once again that routine water monitoring exercises will not typically detect levels of neonics as high as those reported in the literature. Data collected as part of broad water monitoring exercises are typically 'grab samples'. As aptly demonstrated (e.g. Xing et al. 2013) peak surface concentrations of pesticides can be underestimated by several orders of magnitude if one relies on grab samples. This point has been made by other authors as well (e.g. Barmentlo et al. 2021); yet the issue is typically not acknowledged by regulators and state authorities who are often content to merely report the fraction of samples that exceed benchmark values. A clear demonstration of this point was the analysis of New York State samples (Mineau 2019b). The following figure, taken from that report shows the relationship between the peak level of imidacloprid detected at any of the water monitoring stations against the number of years there was a detection. The more times a site is sampled, the better the chance to record values approaching true peak events.

Figure 2.1. Maximum imidacloprid concentration (in µg/L) recorded per monitoring site in New York State against the number of years imidacloprid was detected at the site. (reprinted from Mineau 2019b)

There is now incontrovertible evidence that pesticide loadings are a key factor in controlling stream quality as judged by its normal complement of invertebrates, especially the more sensitive groups of macroinvertebrates such as mayflies, caddisflies, and aquatic beetles. (Reiber et al. 2020; Liess et al. 2021). Clearly, as the most important class of insecticides, the neonics are playing an important part in the degradation of freshwater systems worldwide as well as, we would argue, estuarine and inshore marine environments. Associating a specific set of compounds such as neonics with biological outcomes such as insect emergence is not an easy task. Aquatic field studies are difficult and their power to detect effects are generally very weak because of the natural variability in such systems and the difficulty of having a suitable number of replicates. Yet, despite these methodological difficulties, the evidence is building that neonics are having clear impacts on aquatic ecosystems much as is being documented in terrestrial systems.

2.5.4. Increasing evidence of reduced insect biomass and emergence as a result of neonic contamination

In our 2013 report, we had reviewed an unpublished MSc. thesis (Van Dijk 2010) from the Netherlands linking neonic contamination to invertebrate numbers in Dutch canals. This work was eventually published as Van Dijk et al. (2013). Vijver and van Den Brink (2014) criticized the conclusions of the study because the authors had failed to account for other pesticide residues present in the watersheds under study. In a roundabout way, however, Hallman et al. (2014) gave credence to the original Van Dijk work by showing quite conclusively that insectivorous birds had declined in response to neonic concentrations (imidacloprid in those days) in water and that the data did not show these declines prior to the introduction of neonics, despite a plethora of other insecticides of high aquatic toxicity. Hallman et al. were able to predict that regional bird declines would begin at water levels of imidacloprid of 0.2 µg/L or higher.

Nowell et al. (2017) were able to show a relationship between mayfly abundance and maximum imidacloprid concentrations from a sample of US Midwest streams. A dramatic paper in Science (Yamamuro et al. 2019) documented the collapse of an entire smelt fishery in Japan as a result of neonic contamination resulting from rice paddy culture. Spring plankton populations declined by 83 percent and the smelt harvest declined from 240 to 22 tons. Total neonic concentration measured in a tributary to the lake was measured at 0.072 µg/L in June of 2018. Three neonics were detected in the lake proper following rice planting in the watershed: imidacloprid, clothianidin and thiamethoxam.

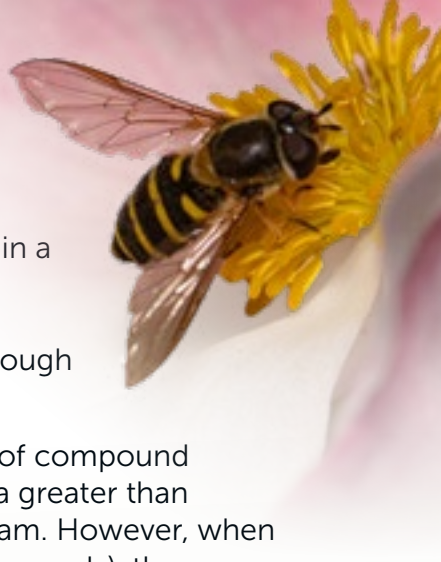
Cavallaro et al. (2019) emphasized the point that wetlands are subject to a number of pressures already, especially in agricultural landscapes, where inputs (e.g. fertiliser, sediment runoff etc.) as well as edge disturbance can dramatically affect aquatic quality and prevent these waterbodies to achieve their full biological potential. Yet, these authors found that neonic inputs (primarily clothianidin and thiamethoxam but also some imidacloprid and acetamiprid; summed residues corrected by a measure of toxic equivalency), affected insect emergence as well as measures of habitat quality and diversity[47]. Their results emphasised a point we made earlier: 73% of their samples had mixtures of neonics from the various canola treatments used over the years.

Schepker et al. (2020) surveyed 26 wetlands within an agricultural area of Nebraska, USA during the spring of 2015. The survey was designed to coincide with the waterfowl spring migration rather than any agricultural activities. The survey therefore reflected pre-planting conditions. Invertebrate communities were assessed between February and April. Nevertheless, imidacloprid (max concentration of 0.005 µg/L) and/or clothianidin (max concentration of 0.016 µg/L) were detected in 85 percent of the sampled wetlands although at levels below the EPA online benchmarks (table 2.1). The authors did find that the presence of a greater than 50m buffer around the wetlands reduced the concentrations of insecticides that were measured[48]. Despite the levels being so low, the authors still saw a negative effect of total neonic concentration on the nektonic biomass – all species confounded.

Barmentlo et al. (2021) applied two biweekly spikes of thiacloprid at concentrations ranging from 0.1 to 10 µg/L to a series of experimental ditches. Odonata (dragonflies and damselflies) as well as trichoptera (caddisflies) proved most sensitive, reduced emergence being clearly seen following the two 0.1 µg/L spikes. Effects on total biomass and diversity were clearly seen following the 1 µg/L spikes. Over a 30-day period, the two 1 µg/L spikes amounted to a time-weighted concentration of 0.3 µg/L. They commented that changes in individual species within a taxonomic order often hid the disruption caused by the insecticide. Some species benefitted from the competition release as the more sensitive species were impacted. As the authors commented, all of the observed changes were at levels of neonics commonly recorded worldwide. They also pointed to the fact that, in part because of the short study duration, their results probably underestimated the full impact that thiacloprid was having on aquatic communities and the insectivorous bird species that depend on them.

[47] Simple metrics such as the number of emergent insects often does not tell the entire story. Species more tolerant of pesticide and other agricultural inputs (i.e. the rats or cockroaches of invertebrate communities) come to dominate an aquatic system and may be present in greater numbers.

[48] A recent, more rigorous analysis of vegetated buffer strips (Vormeier et al. 2023) argued that the most effective method of reducing stream impacts from pesticides was to institute an 18m buffer. However, if ditches are present along the fields or roads, the vegetated buffer strip needs to be doubled in order to gain the same protection benefit. This was for German cropping and weather conditions.



2.5.5. Additivity or synergisms

We argued in our earlier report, that a compound by compound approach as currently adopted by American and Canadian regulators no longer makes any scientific sense given the frequency with which multiple residues are detected in a variety of aquatic ecosystems.
Morrissey et al. (2015) similarly advocated looking at summed residues and, as reviewed earlier made the case that toxicity benchmarks were close enough that a joint toxicity benchmark was possible.

Maloney et al. (2017, 2018 a, b) obtained contradictory results on the principle of compound additivity. Laboratory results on a chironomid species seemed to clearly show a greater than additive effect with combinations of imidacloprid, clothianidin and thiamethoxam. However, when the experiments were taken out of doors in a series of pond mesocosms (limnocorrals), there was no evidence of synergistic effect between compounds. However, impacts on chironomid emergence were generally greater than would have been predicted from laboratory data but high variability among pond replicates makes interpretation difficult. It is intriguing that Bayer Corp., one of the major manufacturers of neonicotinoids had argued that several neonicotinoid insecticides could act synergistically and had obtained a patent on this finding (Andersch et al. 2010).

In a recent ground-breaking study published in Science, Schmidt et al (2022) were able to combine field observations in 85 coastal California streams with mesocosm testing of the two dominant neonics: imidacloprid and clothianidin. Mayfly abundance (all species combined) was clearly affected by both imidacloprid and clothianidin. A 50 percent reduction in abundance was seen at time weighted average concentrations (over 30 days) of 1.05 µg/L and 1.35 µg/L for imidacloprid and clothianidin respectively. However, examination of the plotted cumulative emergence over time appears to show clear effects at concentrations as low as 0.001 µg/L with either clothianidin, thiamethoxam and to a lesser extent, imidacloprid (Figure 2 in Schmidt et al. 2022). That is a level 10 times lower than the older more protective EPA chronic benchmark for imidacloprid, 125 times lower than the new benchmark being used by EPA (USEPA 2022f) to assess risk to endangered species. Adding their test data to the existing corpus of chronic studies, the authors derive chronic HC_5 values of 0.017 µg/L for imidacloprid and 0.010 µg/L for clothianidin. The fact that these chronic HC5 values may not be low enough to maintain cumulative emergence of mayflies argues for the need to reconsider how we look at neonic toxicity – see the earlier discussion on time-weighted toxicity above.

Through their series of experimental streams (mesocosms) the authors were able to confirm that imidacloprid and clothianidin were behaving in a greater than additive fashion – they acted synergistically in many cases. The authors found, in their field samples, that total extirpation of mayflies occurred at concentrations of imidacloprid or clothianidin that only caused a 50 percent decline in abundance with one or the other compound in a mesocosm setting.

Neonic mixtures were detected in 56 percent of streams. At least one neonic was detected at 72 percent of the sampled streams (N=85). Summed neonic residues were as high as 5.76 µg/L. Imidacloprid was often the dominant compound in the mixture but dinotefuran was the most frequently detected and thiamethoxam was recorded at the highest concentration. The authors reported that at least one of the EPA benchmarks (the online levels in table 2.1) were exceeded in 28 percent of the samples. All samples were taken in April-June 2017 under low-flow conditions and might therefore not have picked up peak residue levels that follow rains although they did cover the time of year when larval communities are well developed.

3. Putting it all together – Direct and Indirect effects on birds

3.0. The unwillingness of regulators to fairly assess neonics and reduce their ecosystem-wide impacts continues to baffle. Credible region- or nation-wide analyses now link declines in bird populations to the use of neonic insecticides. An unprecedented 2023 Europe-wide analysis shows that agricultural intensification, as measured by the use of pesticides and fertilizers, is the main driver of bird population declines. The direct link between the inputs associated with intensive agriculture and bird declines is clear. The role of neonics in the intensification of insect control is just as clear.

We are now at the point where the onus of proof should switch from having to demonstrate the link between neonicotinoid use and bird populations losses as well as environmental degradation of aquatic systems, to showing why the continued profligate use of neonics is essential to human welfare in light of such environmental impacts. Regulators in North America claim to objectively review the available science and use weight of evidence approaches. Unfortunately, we believe they have failed in the execution of their mandate and in preventing the ongoing environmental tragedy that neonics represent.

The most troubling aspect of the regulatory saga of neonics is the insistence of regulatory bodies (whether the EPA or PMRA in Canada) to ignore the increasing field evidence of impacts while playing with small permutations of their risk calculations in favour of industry. In its most recent assessment, the EPA (2022f, g, h) carries out its risk assessment as if neonics were new introductions to the market; despite reviewing the scientific literature, there is no evidence that any of it is considered in the risk assessment proper. Ignoring the growing evidence of real-world impacts is not what is meant by science-based decision making.

Notable are the analyses of Hallman et al. (2014) for Dutch insectivores and Ertl et al. (2018) for quail species in Texas making a very convincing case for bird declines in response to the presence of neonics in their environments. The first argued for an indirect effect through water contamination; the latter for direct effects following exposure to treated seed. Indeed, as argued in this and our previous 2013 report, both mechanisms are possible and likely.

Li et al. 2020 published an amazing analysis in Science. They provided an analysis of Breeding Bird Survey data, linking significant farmland bird declines to neonic use. They were able to show that the results were due to neonics and not other pesticides (with similar but lesser effects). They were also able to separate the effects of neonics from habitat and other components of agricultural intensification (which did have an impact, but again of lesser magnitude) or weather variables. They separated data into insectivorous vs. non-insectivorous species and then into grassland vs. non-grassland species. They were able to attribute reductions in bird numbers to all groups studied with equal effects on insectivorous vs. non-insectivorous species. Of interest is their finding that neonic use, once weighted by bird toxicity was shown to have twice the effect of the non-weighted use. Those two findings, especially the weighted analysis, suggest that direct toxicity to birds is a clear factor in the declines. The fact that there were equal effects on insectivorous vs. non-insectivorous species does not rule out an effect through insect removal however, because all bird species, whether granivorous or insectivorous, feed insects to their young. In grassland species, especially, toxicity weighted neonic use affected abundance, species richness and species evenness, the latter measure indicating that some species were harder hit than others.

To properly assess the importance of neonics on bird populations, Li et al. (op. cit.) looked at both the immediate effects of neonic use, and designed a dynamic model which considered the lag effect resulting from there being fewer birds to reproduce and give rise to the next generation. They found that the actual increase of neonic use between 2008 and 2014 (seed treatments are no longer

accounted for in use statistics beyond 2014) had resulted in a yearly loss of between 4 percent (static estimate) and 12 percent (dynamic estimate) of grassland species numbers nation-wide; those estimates were 3 percent-5 percent for insectivorous species. On a county by county level, they were able to estimate that every 100kg increase in neonic use resulted in a 3.5 percent decline (+/- 1.4 percent) in insectivorous birds.[49] In the few counties where neonic use actually declined during this period, they were able to see a small rebound.

Lennon et al. (2019) had not found similar bird declines in the UK. The main neonic uses in that country were seed treatments for spring crops (primarily sugar beet) in the first part of the study period (1994-2004); while the application to fall-seeded oilseed rape and cereals dominated in the second part of the study period (2005-2014). It should be noted that, by North American standards, this represents a relatively small intensity of use. Where neonics were used, the median estimated application rate was calculated to be 0.0001 kg of imidacloprid-equivalent neonic use per acre and the maximum 0.0113 kg of imidacloprid equivalent use per acre [converted from hectares to acres by authors]. To consider the impact of fall-seeding, the authors looked at the breeding bird data for the subsequent year. They saw no consistent effect of neonic seed treatments on breeding bird numbers sampled at the 5 km X 5 km scale even after applying weightings to account for the differential toxicity of the different neonics, or dividing the farmland bird species into low, medium or high exposure groups based on their susceptibility of exposure by factoring in dietary preferences. There were several species in the high and medium exposure groups that had positive relationships to neonic use despite the fact that they were generally declining in the UK during that time. Only four species showed significant negative estimates associated with neonic use: the Turtle Dove (*Streptopelia turtur*) (noted as a high exposure species) and House Sparrow, Red-legged Partridge and Skylark (*Alauda arvensis*) (noted as medium exposure species). In the case of Turtle Doves, although they were judged to be at high exposure risk, their migratory habits meant they had mostly departed at the time of autumn seeding.

Effects on birds can be complex and difficult to unravel. A case in point is Sigouin et al. (2021). As part of a long-term study of Tree Swallows in agricultural areas of southern Quebec, they found a synergistic effect between pesticide exposure and parasitism. When exposed to heavier pesticide loads (as measured by contamination of food boluses brought to the young) nestlings were less able to compensate for the blood loss resulting from parasitic fly larvae resulting in significantly-reduced haematocrit levels. Two other blood markers associated with immune response were not affected. Although a large number of pesticides were measured, four neonicotinoids were present among the top dozen active ingredients detected: clothianidin (the most frequently detected insecticide), followed by thiamethoxam, imidacloprid and thiacloprid. In a companion study (Poisson et al. 2021) looked at the effect of chick exposure on reproductive parameters. They did not find that the proportion of contaminated boluses affected reproductive parameters however.

As mentioned earlier, it is clear that an abundant and nutritious insect supply is critical to bird guilds such as aerial insectivores and it is clear that there are multiple examples where pesticide use has been shown to disrupt that food supply (Spiller and Detmers 2019). Tallamy and Shriver (2021) recently reviewed the evidence for a link between insect declines and diminishing numbers of insectivorous birds in North America as a whole.[50] They review the numerous studies that have related chick condition (a good predictor of survival to fledging) with the availability of an abundant

[49] The estimate was a 4.9% (+/- 1.9%) decline in insectivorous birds when the data were analysed by crop-reporting district. Calculated population declines were similar for all grassland species.

[50] A similar relationship has been postulated for bats (Mineau 2019).

and nutritious insect supply as well as the growing number of studies that link insect availability to regional population size. They simply (sometimes, simple is sufficient) divided terrestrial bird population estimates into those for which insects are essential (304 species) and those for which they are not (64 species). The bulk of bird declines over the last 50 years have been for species denoted as needing insects, at least for part of their life-cycle. They report a 111-fold difference between the two groups – loss of an estimated 2.9 billion birds for 'insect obligates' versus a gain of 26.2 million birds for the 'non-insect obligates'. The authors do not separate the species into agricultural and non-agricultural species and, without apparently accounting for the evidence presented in this report, argue that plant-insect dynamics, namely the replacement of native vegetation by introduced species and the inability of the latter to yield healthy phytophagous insect populations, may be to blame.

Finally, an unprecedented Europe-wide analysis by Rigal et al. (2023)[51] documents that of the four main anthropogenic stressors (agricultural intensification, forest cover, urbanization and temperature), agricultural intensification as measured by the use of pesticides and fertilizers, is the main driver of bird population declines. The direct link between the inputs associated with intensive agriculture and bird declines is now clear. The role of neonics in the intensification of insect control is just as clear.

Clearly, we cannot impute the totality of past and current bird losses to insecticide use, let alone neonics. However, because pesticides and neonicotinoids especially have been linked to widespread invertebrate declines both in terrestrial and aquatic environments (see above), and because the insect-killing power of current insecticides (dominated by neonics in most of the world) has increased so much over the past decades (e.g. Goulson et al. 2018; DiBartolomeis et al. 2019), we believe that we can go further than Tallamy and Shriver (op. cit.) when it comes to assessing the relative contribution of insecticides and neonics especially. Clearly, population declines are the result of several factors. However, pesticides rank highly and consistently as a factor in those declines (Stanton et al. 2018 and this report). Even bird species which should be thriving in an agricultural environment with all the food resources it offers are declining. Notwithstanding the data of Lennon et al. (2019), the weight of evidence to date supports the view that neonic use is affecting bird populations both directly through lethal and sub lethal poisoning and indirectly through their food supply.

We firmly believe that we have now reached the point where the onus of proof should switch from having to demonstrate the link between neonicotinoid use and declines in bird populations or the degradation of habitats they rely on, especially aquatic systems, to showing why the continued profligate use of neonics is essential to human welfare. Regulators in North America claim to objectively review the available science and use weight of evidence approaches. Unfortunately, we believe they have failed in the execution of their mandate and in preventing the ongoing environmental tragedy that neonics present. This is despite the fact that there is a broad consensus among independent scientists and enlightened regulators; viz. European Academies Science Advisory Council (EASAC 2015), United Nations General Assembly (UN 2017), Worldwide Integrated Assessment of Systemic Pesticides under the International Union for the Conservation of Nature (Furlan et al. 2018), Institut du Développement Durable et des Relations Internationales (IDDRI 2018), European Commission (2022) etc... that we need a major policy shift that moves us away from profligate use of pesticides, especially those, such as neonics, that combine broad-spectrum toxicity to vertebrates, terrestrial and aquatic invertebrates with high persistence and high environmental mobility characteristics. American and Canadian regulators at the federal level truly appear to be operating in a vacuum.

[51] The sheer scale of this analysis is mind-boggling with 52 authors and data from 28 European countries.

4. Regulatory responses and lack of action in North America

4.1. Neonicotinoids in the European Union and Canada

4.1. In the European Union and a few Canadian provinces, neonicotinoids are much more heavily scrutinized and regulated. The EU, especially, has taken major steps in banning outdoor uses of neonics and, more recently the emergency uses of neonics were overturned in a court, fully banning their use. Canadian provinces have taken a more surgical approach recently by requiring specific prescriptions for neonic use on corn seed, leading to a massive decline in their use.

Though the United States remains markedly behind in pesticide regulation, Canadian provinces and the European Union are taking greater action in response to reported impacts from the use of neonicotinoids.

4.1.1. State of Neonicotinoid Use in the European Union

Fortunately for European avifauna, the EU imposed several bans on the main neonics in 2018 (European Commission 2018). The bans were the result of years of noted decline in insect pollinators such as bees and butterflies, as well as hundreds of other species. Impacts on birds were noted as being of particular concern with regard to neonicotinoids, though they were not seen as a primary driver of the regulations.

If anything, the European Union has seen this action (then painted as extremist) as inadequate for facing the current plague of pollinator declines within the continent. On June 22, 2022, the European Commission announced its intention to reduce the use and risk of chemical pesticides by 50 percent by 2030 (European Commission, 2022). This announcement came in close proximity to new rules on chemical pesticides which were intended to "*reduce the environmental footprint of the EU's food system, protect the health and well-being of citizens and agricultural workers, and help mitigate the economic losses that we are already incurring due to declining soil health and pesticide-induced pollinator loss.*" (European Commission 2022)

A component of the 2018 decision was that thiamethoxam, clothianidin, and imidacloprid remained available for emergency uses until 2023, particularly for use on sugar beets. Sugar beet growers and sugar producers have stated that no viable alternatives to neonicotinoids exist in controlling weevils, flea beetles, and aphids on sugar beets. Despite these concerns, a lawsuit challenging the increasing use of emergency exemptions was launched in 2019 and ruled upon in January of 2023. The court ended the emergency uses of neonicotinoids, finding that they violated the 2018 ban (European Court First Chamber 2023). Time will tell how this will impact sugar beet and sugar production in the EU, if at all.

A recent (February 2, 2023) news release from the European Union announced their intent to reduce maximum residue limits of clothianidin and thiamethoxam "*to the lowest level that can be measured with the latest technologies*" (European Commission 2023). Once implemented, these new restrictions will apply not only to products of the EU but also to all imported food and feed. It will be interesting to see whether these European regulatory decisions can provide a roadmap for future action in the United States or Canada. In the opinion of the authors, we feel it is a much-needed and long-overdue action to take.

4.1.2. Neonicotinoid Seed Coatings in some Canadian Provinces

Neonicotinoid-limiting regulations are not unique to Europe. In July of 2015, the province of Ontario, Canada instituted new requirements mandating corn and soybean farmers (where the corn

is grown for grain or silage and the soybean is not grown for certified seed) to obtain prescriptions for using imidacloprid, clothianidin and thiamethoxam as a seed treatment prior to use(Ontario Government 2015). Overall pollinator incidents have severely declined since these and other mitigation measures were introduced (Health Canada 2020).

Pesticide use reports from the province of Quebec, which instituted required neonicotinoid-coated seed prescriptions (obtained from a registered agronomist) beginning in 2019 (Québec Environnement 2018), show a precipitous decline in the use of neonicotinoids as seed treatments in the resulting years. Corn and soybeans, two of the crops which historically used neonicotinoid seed treatments more than any others, have gone from nearly 100 percent of acres treated to almost 0 percent (corn), and roughly 50 percent of all acres to 0 percnet (soybeans).

"*There are virtually no retail sales of seeds coated with neonicotinoids. It is estimated that less than 0.5% of the corn area is planted with a neonicotinoid seed treatment, compared to 100% in 2015. The entire soybean crop is now planted without neonicotinoids, compared to 50% in 2015.*" (Translated from Québec Environnement 2023).

Since the requirement of a prescription for neonicotinoid-coated corn, soy, and canola, there has been a 70 percent reduction in neonicotinoid use in Québec as measured by the amount of active ingredient (Table 4.1).

Table 4.1. Table 5 in Québec Environnement 2023 showing neonicotinoid retail sales between 2018 and 2021.

Ingrédient actif visé par une prescription agronomique	Ventes au détail (kg i.a.)			
	2018	2019	2020	2021
Atrazine	35 797	19 862	5 579	5 098
Chlorpyrifos	–	5 943	5 558	4 047
Clothianidine	–			
Application foliaire	–	2 030	1 543	945
Enrobant Jes semences	–	257	151	159
Thiaméthoxame	–			
Application foliaire	–	774	1 260	1 064
Enrobant Jes semences	–	309	14	8
lmidaclopride	–			
Application foliaire	–	231	101	182
Enrobant Jes semences	–	0	0	0
TOTAL	**35.797**	**29 406**	**14 206**	**11 503**

In just eight years, after requiring a specific prescription for the application of a neonicotinoid on a seed, use of neonics on corn and soybeans dropped from 100 percent and 98 percent to 0.2 percent

and 1 percent respectively[52]. Despite early voiced concerns, there is no major shortage of corn or soybeans being reported in the provinces, showing that when a demonstrated need is required before the use of a neonicotinoid-coated seed, they are rarely required on large-scale row crops. As is clearly shown, the greatest use of neonicotinoid coated seeds is largely unnecessary.

4.2. Neonic Seed Coating Regulations in the United States

4.2. The most common use of neonicotinoids in the United States is as a seed coating; yet due to a loophole in federal regulatory law, they are regulated (or, not regulated) as exempt pesticides. Entitled the Treated Article Exemption, this gap in the law allows voluminous use of neonics without proper oversight from the Environmental Protection Agency. The lack of adequate coated seed oversight has led to an underrepresentation of impacts on wildlife and communities from neonicotinoids. Highly-regarded and relied upon sources of information, such as the NAWQA pesticide use maps, make specific note that seed coatings are too difficult to reliably source information on and, therefore, are not included in national pesticide-use estimates. Despite the dearth of information, there is not a perceived need by federal regulators to reclassify them.

Neonicotinoid seed coatings remain a major part of agriculture in the U.S. despite growing evidence of their lack of efficacy or economic benefit (USEPA 2014, Li et al. 2022, Pennsylvania State University Extension 2023 ... and several other references given in the Introduction to this report) and severe impacts on non-target wildlife and ecosystem function (as reviewed throughout this report). The EPA maintains that the registration reviews of active ingredients are more than adequate to assess the impacts of seed-coatings on non-target organisms and the environment. The devastation wrought by neonicotinoids and neonic-coated seeds certainly argues otherwise.

4.2.1. The FIFRA Treated Article Exemption and pesticide-coated seeds

Under the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), the law which mandates review of active ingredients during registration and registration-review in the United States, pesticide-coated seeds currently are not seen as "pesticides" but rather as "exempt pesticides" via the Treated Article Exemption (TAE) (US National Archives Code of Federal Regulations 2023 - 50 CFR §152.25(a)).[53] The Exemption reads as follows:

> *The pesticides or classes of pesticides listed in this section have been determined to be of a character not requiring regulation under FIFRA, and are therefore exempt from all provisions of FIFRA when intended for use, and used, only in the manner specified.*
>
> a. ***Treated articles or substances.*** *An article or substance treated with, or containing, a pesticide to protect the article or substance itself (for example, paint treated with a pesticide to protect the paint coating, or wood products treated to protect the wood against insect or fungus infestation), if the pesticide is registered for such use.*

[52] It should be noted that atrazine and chlorpyrifos were also subject to similar requirements to obtain an agronomic prescription and these pesticides also saw a marked decline showing the effectiveness of this type of regulatory measure. Chlorpyrifos sales dropped by 66% but the decline occurred in 2017 and is therefore not captured in the table.

[53] In an effort to harmonize with the U.S., the Canadian PMRA has adopted the same position (PMRA 2003).

The Exemption was intended to prevent products which contain a pesticide for the express protection of the item itself from being registered as a pesticide. As seen above, this includes inanimate objects like paint and lumber, whereby those articles may contain a pesticide but are not considered pesticides per se and do not need to be subject to registration as a pesticide themselves. Unfortunately, this has been inappropriately applied to pesticide-coated seeds wherein the seed is coated with an active ingredient (such as a neonic) with the intention that the coating should slough off, be absorbed by the growing plant, and then be incorporated into plant tissues to protect against pests. As reviewed earlier, neonicotinoids are systemic, meaning that the material which is initially present as a seed coating is subsequently incorporated into nearly all living plant tissue. In addition, as reviewed in this report also, a large proportion of the seed coating material is subject to abrasion and escapes as dust to be deposited off site. In no reasonable fashion could this be interpreted as the satisfying the goal of the TAE; unlike other products included under the Exemption (lumber, paint, etc.), coated seeds become plants which are orders of magnitude larger than the original coated product, increasing their potential negative impact exponentially.

Further, 50 CFR §152.25(a) states that the pesticide must be used to protect "*the article or substance itself*" and not resulting products. Plainly, this does not include crop plants which are substantively different from the seed from which they originated. However, in decisions and responses discussed in the following sections, EPA has indicated that the resulting seedling is *also* part of the "seed" being treated, and therefore allowed to be included under the exemption.

> *"Thus, EPA reads this regulatory text to similarly apply to the article treated, i.e., the seed, and the treated article in use and what the article becomes, i.e., the seed after it is planted." (USEPA, 2022b).*

The Exemption was written into law well-before the explosion in the use of neonicotinoids as seed coatings, and despite repeated objections by non-governmental organizations, and against the advice of the State 'FIFRA Issues Research and Evaluation Group'[54] (AAPCO 2022), EPA maintains the status quo to this date.

4.2.2. Petitions to EPA to close the Treated Article Exemption Loophole for Pesticide-coated Seeds and Demonstrate Efficacy

In 2017, the Center for Food Safety and numerous other individuals and organizations including the American Bird Conservancy (ABC), the American Beekeeping Federation, and several growers, petitioned the EPA to close the loophole created by the TAE (Center for Food Safety, 2017). The petition sought:

> *"... an amendment to, or a formal re-interpretation of, the Treated Article Exemption, 40 C.F.R. §152.25(a), to clearly communicate to the regulated community that systemic pesticidal seeds intended to kill insect pests of the plants are not included under the Treated Article Exemption and are therefore subject to FIFRA's requirements. Petitioners also request that EPA aggressively enforce FIFRA's registration and labeling requirements for each separate seed product coated with a systemic insecticide."*

[54] Group of state lead agencies on pesticides. FIFRA is the 'Federal Insecticide, Fungicide, and Rodenticide Act', the federal U.S. statute that governs registration, distribution, sale and use of pesticides.

After receiving the petition, EPA opened a comment period until March 16, 2019, during which it received more than 16,000 comments. Despite clear interest and engagement from stakeholders and the public, there was no action taken.

In 2021, after failure to act on the 2017 petition, the Center for Food Safety and Pesticide Action Network of North America sued the EPA under the Administrative Procedure Act for failure to comply with the accepted petition from 2017 (Center for Food Safety, 2021). Pursuant to a settlement approve by the District Court for the Northern District of California EPA agreed to respond to the petition by fall 2022. (USEPA, 2022a).

The resulting Petition Denial maintained that EPA felt it was adequately assessing pesticide coated seeds and did not need to reconsider their status under the Treated Article Exemption (USEPA, 2022b). Similarly, ABC sent its own letter expressing its displeasure with the finding; this elicited a sympathetic response from EPA but its previous decision was upheld. ABC and partners maintain their opposition to EPA's position on the issue.

Public Employees for Environmental Responsibility (PEER) and ABC, in conjunction with partners, filed a petition with EPA in February of 2023 urging that systemic insecticides, such as neonicotinoids and organophosphates, must require proof of efficacy before being registered. Currently there is no technical requirement for a pesticide registrant to demonstrate proof of their chemical or its use having the required effect. EPA does often request this data during registration or registration review of active ingredients; however much of the data submitted remains opaque due to protection of intellectual property and proprietary industry information of registrants. The petition would add an amendment to 40 CFR § 158.400(e)(1), the section of the US Code governing EPA Pesticide Registration, as follows:

> *"Each existing registrant of a neonicotinoid or other systemic insecticide who has not already submitted efficacy data must submit data on whether its product is efficacious within 180 days of the promulgation of this Rule, whereupon the Agency will consider the product's foreseeable benefits and costs to the environment. The Agency shall not register, and shall revoke any existing registration for, any neonicotinoid or other systemic insecticide that lacks a demonstration that its benefits exceed its environmental and overall costs." (Public Employees for Environmental Responsibility et al. 2023).*

4.2.3. Tracking of Neonicotinoid-Coated Seeds in the United States with the USGS NAWQA Mapping Project

The United States Geological Survey (USGS) acquires commercial surveys[55] and investigations into current pesticide use in the United States as part of the National Water-Quality Assessment project (USGS 2019). These surveys are then compiled and presented publicly via the Pesticide National Synthesis Project as maps and graphs. These maps, and the data they are based upon, are important tools for researchers, regulators and legislators.

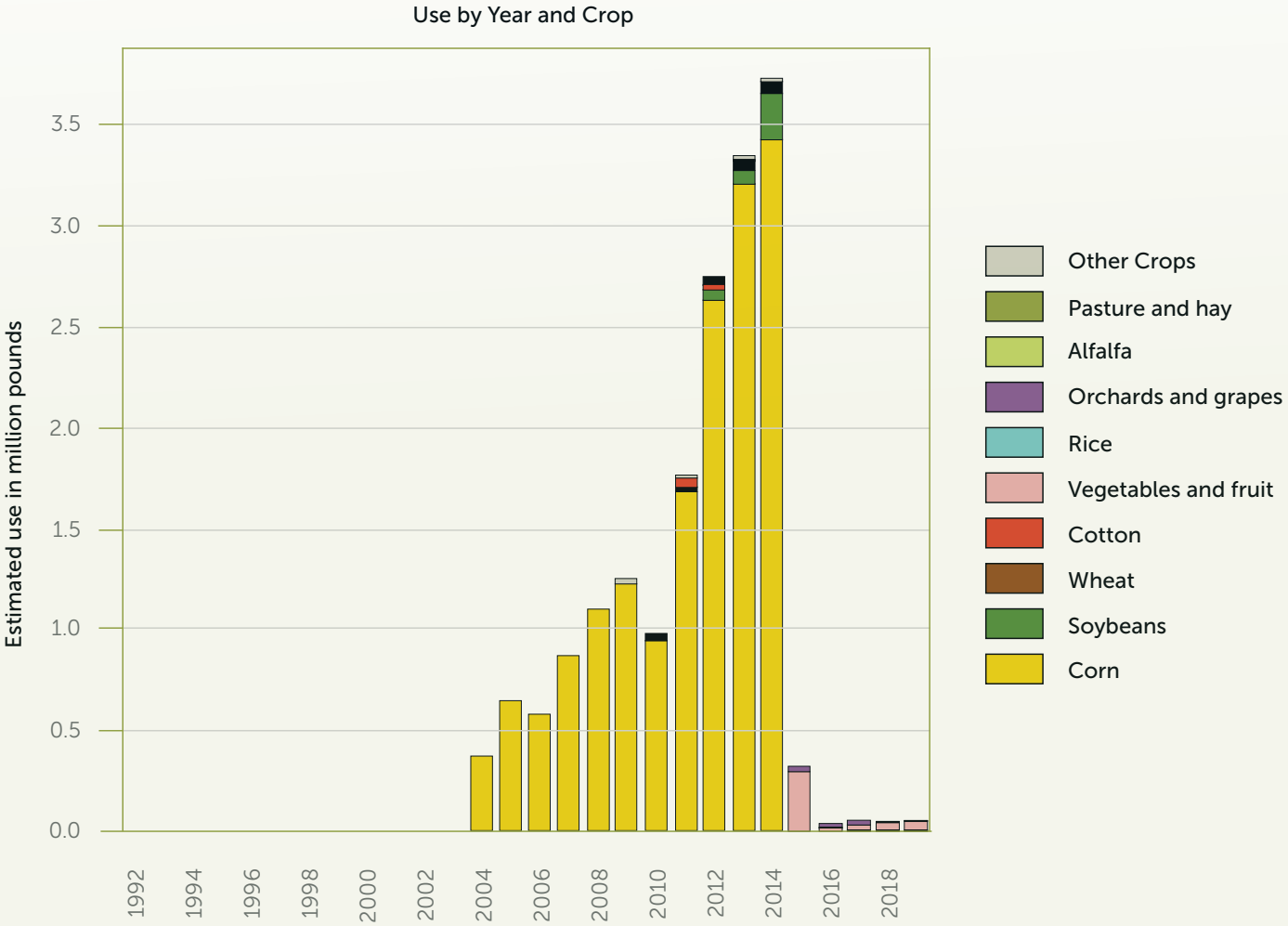
Among the chemicals inventoried are the neonicotinoids acetamiprid, imidacloprid, clothianidin, thiamethoxam, and thiacloprid. Until 2015, pesticide surveys and subsequent data included estimates of use as seed treatments. According to the website:

[55] For example, surveys performed by the firm Kynetec.

"Beginning 2015, the provider of the surveyed pesticide data used to derive the county-level use estimates discontinued making estimates for seed treatment application of pesticides because of complexity and uncertainty. Pesticide use estimates prior to 2015 include estimates with seed treatment application." (https://water.usgs.gov/nawqa/pnsp/usage/maps/)

The 'uncertainty' referred to in the USGS quote is that, increasingly, growers are not even aware that they are applying a pesticide to their field when they are seeding (Hitaj et al. 2020). Consolidation of the seed and pesticide industry as well as the Treated Article Exemption documented above has allowed pesticide manufacturers to sell (and charge the grower for) their neonic insecticides when these are not needed in most cases. The current exclusion of seed treatments from USGS compilations gives the false impression that neonicotinoid use has drastically declined, as exemplified in Fig 4.1. taken from the USGS website. Numerous other chemicals present similarly on the website.

Figure 4.1. Graph of Clothianidin from water.usgs.gov. Due to difficulty in tracking treated seed use, the graph may give the impression that use drastically declines beginning in 2015, however the precipitous drop is only from lack of reporting use as seed treatment.



4.3. Current Impacts of Neonicotinoids on Threatened and Endangered Species

4.3. When EPA does not properly assess the toxicity of a pesticide to exposed species and does not quantitatively account for most of that pesticide's use, how can it possibly arrive at the correct conclusions? The threatened and endangered species assessment carried out by the EPA represents a monumental effort. However, the assessments of neonic effects on TES are highly under representative of true impacts. It is unfortunate indeed that on both the terrestrial and aquatic side, fundamental errors in EPA's assessment methodology renders the assessment unreliable and under-protective of T&E species.

In April 2022, EPA released a first-of-its-kind document, a plan for assessing and mitigating the impacts of pesticide use on threatened and endangered species and their critical habitat. Entitled "Balancing Wildlife Protection and Responsible Pesticide Use: How EPA's Pesticide Program will meet its Endangered Species Act Obligations" (USEPA, 2022c).

The plan includes acknowledgement of EPA's historic lackluster response to its Endangered Species Act obligation to consult with the U.S. Fish and Wildlife Service and National Marine Fisheries Services during pesticide registration. The plan was preceded by an announcement in January 2022 that all new active ingredients would receive consultations for impacts on TES (USEPA, 2022d), and was followed by an update to the plan in November 2022 which outlined specific strategies for mitigating pesticide risks to nontarget species and other action (USEPA, 2022e).

Neonicotinoids were included in these documents as active ingredients undergoing review and evaluation driven by litigation. In June 2022, the final biological evaluations for clothianidin, imidacloprid, and thiamethoxam (2022f, g, h) were released. Some of the limitations and errors in these assessments have been reviewed extensively in this report. Nevertheless, the assessments concluded as follows with respect to species thought to be likely to be adversely affected (LAA) by current registered uses of the active ingredients (Table 4.2).[56]

Table 4.2. Percentages of threatened and endangered species and their habitats deemed likely to be adversely affected by the first-generation neonicotinoid insecticides (USEPA 2022f, g, h).

Active Ingredient	% TES LAA	% TES Bird Species LAA	% CH LAA
Imidacloprid	79	71	83
Clothianidin	67	83	56
Thiamethoxam	77	71	81

TES = Threatened and Endangered Species, CH = Critical Habitat, LAA = Likely to be Adversely Affected

[56] Table created with numbers derived from the documents included in this press announcement: https://www.epa.gov/pesticides/epa-finalizes-biological-evaluations-assessing-potential-effects-three-neonicotinoid

It is clear that the EPA (or at least some scientists within that Agency) does acknowledge seed ingestion (as has been previously documented in this report), and the fact that ingestion of large numbers of seed at spill sites does occur, and that only a few seeds are required to reach toxic thresholds. For example:

> *"Given the potential availability of other seed sources (i.e. remaining waste grain or seeds from weed species on the field), eating diets made up entirely of a specific seed type is unlikely but may be more likely in instances of treated seed spillage than through normal foraging behavior." (USEPA, 2020i).*

However, the final revised methods adopted by the EPA to assess the risk of pesticides to TES do not appear to even acknowledge the presence of seed treatments as a potential route of exposure (USEPA 2020j). Indeed, the assessment of seed treatments takes a very bizarre turn in the 2022 final biological determinations (USEPA 2022 f, g, h). Firstly, the EPA decries the lack of information on seed treatments; for example:

> *"However, quantitative seed treatment usage data are difficult to obtain due to the complexities of capturing this usage information from growers." (USEPA 2022g; Clothianidin Appendix 1-4. Usage Data. 2022).*

What the EPA does not do is acknowledge that this lack of information is largely a result of their insistence on putting treated seeds under the Treated Article Exemption provisions. The EPA goes on to provide an analysis to show which parts of the country could benefit from better use data – a large part of the US (e.g. USEPA 2022f, Appendix 4-5).

In the same analysis, the EPA concludes that a large number of TES would overlap with treated seeds if the proportion of seeded crops using treated seeds was high (viz. appendices 4-5 in the 2022 assessments). We already know as reviewed in this and other reports (e.g. Mineau 2020) that the proportion of treated crops is indeed very high for the most important uses of neonics, approaching 100% in field crops such as corn and soy. Bizarrely, the analysis for birds at least, goes on to not consider most of the species with the highest potential overlap and those most likely to be exposed directly to treated seeds or residues from same: e.g. Whooping Crane, Streaked horned lark (*Eremophila alpestris strigata*) , Yellow-billed Cuckoo (*Coccyzus americanus*), etc[57]...

For most or all endangered bird species assessments carried out by EPA, it appears[58] that the estimate of overlap between a species' range and pesticide use is based, therefore, on soil and foliar applications of liquid formulations of the same active ingredients. An analysis of California's use of neonics (Mineau 2020) showed that, even for states where field crops represent a smaller proportion of the total crop area, not accounting for seed treatments may underestimate use by 50 percent and completely distort the estimated extent of a compound's 'action area' which is used by the EPA to assess risk to

[57] These and other TES were clearly identified as species likely to be affected in previous expert declarations in a successful court challenge from the Center of Food Safety against EPA and Bayer Cropscience L.P. (Mineau 2014, 2015, 2018).

[58] We use this word advisedly because the logic behind the EPA biological assessments is very convoluted and opaque indeed. A small number of 'test cases', all of which bird species show no potential overlap with seed treatment applications even if seed treatments were used on the entire crop modeled (e.g. Hawaiian goose – *Branta sandvicensis*; Puerto Rican Plain Pigeon – *Columba inornata wetmore*; and Puerto Rican Parrot – *Amazona vittata*) are put forward to argue that seed treatment use would not change any species' likelihood of being affected. It then appears that the impact of seed treatments on any bird TES is therefore dismissed as a result of these three species. We would be thankful if anyone could better explain the logic behind these comparisons.

threatened and endangered species. It is therefore clear that any determination resulting from these biological evaluations is not scientifically credible and that the EPA, through a faulty logic model, has greatly underestimated the extent of the risk, for birds and probably other species as well.

In May 2023, EPA took the next step in their analyses and determined which species among those 'likely to be adversely affected' would likely be jeopardized or have their 'critical' habitats adversely modified. Despite severely underestimating the toxicity of the neonics and the extent of overlap between use and species' ranges, they estimated that a staggering 200+ TES species and 30+ critical habitats are currently in jeopardy. Despite the numerous flaws in these assessments as documented throughout this report, and the fact that these numbers are therefore much lower than would be revealed by a competent science-based assessment, this represents a monumental advance in the evaluation and regulation of neonicotinoids and shows EPA's willingness to deal with the issue. The results are summarized below in Table 4.3.[59]

Table 4.3. Percentages of threatened and endangered species and their habitats deemed to be in 'jeopardy' (a legal definition under the Act) or to have their 'critical habitat' adversely affected by the first-generation neonicotinoid insecticides.

Active Ingredient	% TES Jeopardized	% CH Likely to be Adversely Modified	# Bird Species Jeopardized
Imidacloprid	11	3	1
Clothianidin	9	4	1
Thiamethoxam	11	4	1

TES = Threatened and Endangered Species, CH = Critical Habitat

Though between 71 percent and 83 percent of ESA-listed bird species were deemed likely to be adversely affected by these three neonics, only 1 species, the Attwater's Prairie Chicken, was judged to be in 'jeopardy'. Its primary threat is assessed as a lack of habitat and plant food resources resulting from decreased pollinating insect abundance. As reviewed above, even if the EPA had the proper toxicity inputs, the logic that is used to assess the risk of seed treatments to birds makes a proper determination impossible. However, a previous assessment in the context of a successful court challenge to the registration of several seed treatment products (Mineau 2018) showed that direct ingestion of treated seed was a clear issue for the species. This is shown also in section 1.4.2. of this report.

The TES assessment carried out by the EPA represents a major effort – if judged by the voluminous nature of those assessments alone[60]. If we were to assign a grade, we could easily give them an 'E' for 'Effort'. It is unfortunate indeed that on both the terrestrial and aquatic side, fundamental errors in EPA's assessment methodology and logic structure renders the assessment unreliable and potentially dangerous in that it is completely under-protective of T&E species.

[59] Table created with numbers derived from documents in this press announcement: (USEPA 2023b) https://www.epa.gov/system/files/documents/2023-05/ESA-JAM-Analysis.pdf

[60] For example, the clothianidin assessment alone clocks in at 460+ pages of text, and a multitude of spreadsheets.

4.4. State Actions to Limit Neonicotinoid Use

4.4. In the absence of adequate federal regulation, some state governments and pesticide oversight bodies have taken steps to limit the use of neonicotinoids. As of May 2023, ten states have enacted legislation making neonicotinoids restricted-use pesticides, meaning only certified pesticide applicators have access to them. However, many of these state laws and regulations include exemptions for agricultural and other intensive uses of pesticides. In certain cases, such as with Nebraska, specific disposal requirements for neonic-coated seeds had to be legislatively enacted after an ecological disaster.

As a result of the failure of the federal government or agencies to adequately regulate neonicotinoids in all forms, limitation of their use has largely fallen to the States. Due to the idiosyncratic nature of a state's need or uses for neonicotinoids, a patchwork of different regulations and laws have been enacted governing neonicotinoids. The following analysis is not meant to be fully representative of all restrictions on neonicotinoids in the United States and is considered current as of May 2023.

4.4.1. States invoking Restricted Use Pesticide Status

As of the writing of this report, approximately ten states have enacted meaningful neonicotinoid restrictions in the form of making some uses of them "restricted use," meaning only a licensed certified pesticide applicator (CPA) is able to access and use them. Not all states use the same vocabulary though the implications are largely the same.

It is worth noting that almost all of the regulations include large exemptions of one type or another. Typically, these 'restricted use status' bills exempt agricultural uses, seed treatments, indoor pest control, and other uses. Their effectiveness is therefore negligible.

Table 4.3. State legislations placing some restrictions on the use of neonicotinoid insecticides at writing.

State	Year Enacted	Bill or Regulation #
Colorado	2023	SB 23-266
Connecticut	2016/2022	SB 231/SB 120
Maine	2022	SB 120
Maryland	2016	SB 198
Massachusetts	2021	MDAR Regulation
Nevada	2023	AB 162
New Jersey	2022	S 1016
New York	2022	DEC Regulation
Rhode Island	2022	H 7129
Vermont	2019	H 205

4.4.2. Disposal requirements in response to an environmental and health disaster

Though pesticide-coated seed bag labels typically include best management practices, these labels are not enforceable in the same way a label on non-exempt pesticides are. This means that proper disposal of pesticide-coated seed is suggested, but not governed, in the same way as other pesticide applications.

As such, an excess of unplanted neonicotinoid-coated corn seed was used in the production of ethanol in Nebraska in the late 2010s. The by-products of ethanol are various corn products known as distillers' grains, corn gluten feed, and corn gluten meal. The plant, AltEn, was using neonic-coated seeds for its ethanol production, rendering the by-products highly toxic due to the nature of the seed treatment chemicals. AltEn had been storing the by-products onsite in the town of Mead, Nebraska. Residents of Mead began reporting nosebleeds, itchiness, and eye irritation from those storage areas. Every bee in the apiary operated by the University of Nebraska at a site one mile from the plant died at a time coinciding with the by-product production (Gillam 2021).

With no federal oversight or potential legal recourse due to the nature of the contamination, the Nebraska state legislature was forced to respond. In 2021, Nebraska enacted LB507, a law which stipulates that neonicotinoid-coated seeds cannot be used for the production of ethanol:

> *"The use of seed that is treated, as defined in section9 81-2,147.01, in the production of agricultural ethyl alcohol shall be prohibited if such use results in the generation of a by-product that is deemed unsafe for livestock consumption or land application."*
> *(Legislature of Nebraska, 2021).*

Other states are considering similar legislation to prevent these types of disaster. The failure to consider pesticide-coated seeds as pesticides clearly leads to inconsistency and danger in the disposal and use of coated seed.

4.4.3. Legislation Specifically Addressing Neonicotinoid Seed Coatings

At present, there are no enacted laws which specifically prohibit seed coatings. A few states, including Vermont and Maine, have legislation which requires the development of best management practices and/or mandates study of the use of pesticide-coated seeds within the state. Ongoing efforts to phase out neonicotinoid-coated seed use should be supported and encouraged at all levels of government.

Conclusion and Recommendations

In an exhaustive review of birds and pesticides, Moreau et al. (2022) commented on the sad state of knowledge when it comes to assessing the contribution of pesticides to bird declines.[61] The sheer number of compounds birds are routinely exposed to, as well as the multitude of ways in which they have been shown to be affected in laboratory settings, argues for pesticides having a large role to play in the worldwide decline of common farmland birds – especially those species that ought to be benefiting from our conversion of native habitat to farmland. Yet, the best that we can do is show correlational relationships between bird numbers and pesticide use. Already, a number of studies reviewed here have shown such correlations for neonics. Do we need more research to be able to clarify the links between specific molecules and bird losses? Yes! Should we, like regulatory bodies in North America, keep ignoring the fact that the current weight of evidence indicates a clear and present danger to birds and other wildlife, as well as the ecosystems they rely on from the current insecticides in widest use? Emphatically no! The current failure of regulators to come to grips with the current environmental impacts of agrochemicals, especially in cases where they are not truly needed, is inexcusable.

More specifically, how can the continued prophylactic use of neonics over millions of acres with debatable benefits to our agricultural systems be justified? Imidacloprid is still registered in North America despite the wealth of information that indicates this neonic is far more directly toxic to birds than the 'second generation' clothianidin and thiamethoxam. This is all the more baffling since it is clear form many jurisdictions that it is not needed because clothianidin and thiamethoxam are often registered as seed treatments for the same crops. Having documented a large number of kills attributable to imidacloprid seed treatments before they were banned in France, Millot et al. (2015) carried out a radio tracking study of Grey Partridges in French farm fields. Imidacloprid had already been banned in France by then and there was very little seed treatment of any kind recorded in their study areas (only six of 186 active ingredients in use were used as granules or seed treatments) – a situation dramatically different from the current situation in North American agriculture. As a result, there was very little direct mortality attributable to pesticide poisoning from any of the pesticides used by their sample of interviewed farmers. Only small amounts of thiacloprid and acetamiprid were reported by those farmers and the only pesticide-related mortality in the study were two carbofuran poisonings.[62] Clearly, the lethal risk to birds in agriculture had decreased over time in that country although more subtle effects are still expected from the cocktail of in-use pesticides (Gaffard et al. 2022). Why can we not follow suit in North America, especially when birds have been given unprecedented protection under the law (Mineau 2004)?

Removal of imidacloprid (and the equally toxic acetamiprid), although a necessary step to prevent many direct bird poisonings, would not be sufficient on its own to right the past decades of regulatory incompetence and inactivity. The demonstrated impact of the neonic class of insecticides on aquatic systems and the broad contamination of the environment resulting from all of these products should be sufficient to restrict them for all but the most essential uses. Errors of omission and commission on the part of North American regulators continue to mount. Despite concluding that a large number of bird species are likely to be adversely affected, the EPA, (USEPA 2023b) now concludes that only one bird species is likely to be placed in jeopardy as a result. This report shows that fundamental issues with

[61] This was before the Rigal et al. (2023) analysis reviewed above.

[62] This is a banned substance in France which is recognized as leading to widespread bird mortality wherever it is used (e.g. Richards (ed.) 2012)

the risk assessment on multiple levels cast serious doubts on this conclusion. Some of these issues were already outlined in our 2013 report. They continue to be ignored, and indeed were compounded by more errors, 10 years later.

Neonicotinoids are used at a scale not seen since the introduction of DDT, yet the regulation of these chemicals is severely lacking. Their most voluminous use, as a seed coating, goes mostly unregulated by federal and state entities in any meaningful way. The United States, as is often the case with pesticide regulation, is distantly trailing the European Union and a few Canadian provinces (Ontario and Québec) in responsible neonicotinoid regulation and mitigation. Alternatives to these chemicals do exist and, plainly, current usage is not justified on agronomic grounds. While the EPA has taken some steps in addressing the issues we raised 10 years ago (e.g. assessment of pollinators, contamination of aquatic systems etc.) these efforts are not moving at a pace sufficient to mitigate the effects of neonicotinoids on wildlife and ecosystems and are fundamentally flawed. Neonicotinoids continue to be registered and used in unprecedented quantities and the environmental damage goes on unabated.

Until pesticide-coated seeds are reclassified as pesticides and removed from the FIFRA 'treated article exemption' and until the EPA addresses the fundamental risk assessment issues we raise in this report, there can be no hope of adequately addressing the risks posed by neonicotinoid insecticides. In the interim, states should continue to pass legislation and/or promulgate regulations reducing the use of these insecticides in order to reduce their sheer environmental impact. Unfortunately, many of the laws and regulations adopted to date will have very little effect because of the numerous exemptions therein. Some have argued that a "one size fits all" approach to state-level mitigation of neonicotinoids is not possible. In fact, the impacts of neonics are largely unavoidable and the same in every jurisdiction. Now even more than in 2013, we see a very strong case for cancelation of all but the most essential uses of neonics.

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