Neonicotinoid Pesticides in Michigan:

Surface Water Contamination and Threats to Aquatic Ecosystems

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1. EXECUTIVE SUMMARY AND CONCLUSIONS

Michigan has seen a massive increase in the use of neonicotinoid pesticides, or "neonics," in the last few decades, driven primarily by the widespread prophylactic use of neonic coatings on crop seeds – known as "seed treatments." As a result, neonics now frequently contaminate Michigan water bodies, likely causing significant and widespread damage to aquatic life.

Neonic seed treatments on corn and soybean seeds are the predominant neonic uses in Michigan, and clothianidin is likely the most used of all the neonics, as it is the dominant insecticide coating on corn seed. The exact amount in use today is difficult to ascertain because data on seed treatment use are no longer collected, but in 2014, roughly 84% of the total agricultural neonic use in Michigan by weight was in the form of seed treatments: primarily clothianidin on corn; imidacloprid on soybeans, wheat, and corn; and thiamethoxam on soybeans and corn. The total agricultural use of neonics exceeded 200,000 pounds of active ingredient (a.i.) in 2014 and was increasing exponentially. Other agricultural neonic uses include ground and foliar sprays in orchards and on grapes as well as on vegetable farms. No information is collected on the use of neonics in other sectors, such as landscaping and structural uses. In corn- and soybean-producing states, these non-agricultural uses typically account for a small fraction of total use, although they do expand the geographic areas where neonics are used – and, as a result, where neonic water contamination is typically found.

Two datasets for water residues were analyzed for this report. The first is from the United States Geological Survey (USGS), which unfortunately tested only for imidacloprid in most cases. Nevertheless, the data show that at all but one of the 22 sites where imidacloprid was detected, it was at levels expected to have aquatic impacts. Sites embedded in the agricultural matrix, such as Randall Drain, a tributary to Saginaw Bay, showed the highest imidacloprid contamination. Levels of contamination were recorded at almost 100 times the current U.S. Environmental Protection Agency (EPA) benchmark for harm to aquatic ecosystems. USGS did carry out more comprehensive sampling of the three main neonics in major rivers over the course of a complete season in 2015–2016. At least two of the sampled rivers, the Grand and the Saginaw, showed yearlong contamination at levels expected to cause ecological impacts.

The second dataset comes from more recent sampling carried out by the Michigan Department of Agriculture and Rural Development (MDARD) and the Department of Environment, Great Lakes, and Energy (EGLE). A total of 54 creeks and rivers were sampled, usually four times during the year. Neonics were detected in half of the sampled rivers, with two or more neonics present at 75% of those sites. Despite missing most peak levels following seeding (sampling was too early in the spring), peak imidacloprid-equivalent concentrations (as described in Appendix A) in those rivers with detections were above the EPA chronic (i.e., longer-term) benchmark for harm to aquatic ecosystems at 77% of the sites where it was possible to calculate a combined residue equivalent. Further, the acute (i.e., short-term) benchmark for harm to aquatic ecosystems recently developed by the European Union (EU), a more scientifically defensible benchmark than the EPA equivalent, was exceeded at more than half of the sites where neonics were detected, at times by a factor of more than 10. This suggests neonics inflict significant and widespread damage to aquatic life in Michigan.

This more recent snapshot of the situation in Michigan also shows that clothianidin, the neonic most associated with seed treatments, dominates most of the samples and accounts for most of the toxic impacts, highlighting the inadequacy of the federal sampling, which largely failed to test for clothianidin residues. There is good evidence to show that clothianidin is about twice as toxic to aquatic life as imidacloprid, the first registered neonic. In this regard, current EPA benchmarks are not sufficiently protective of water resources.

While the available data and current benchmarks already indicate that the expansive prophylactic use of seed treatments poses a significant threat to the health of aquatic life in Michigan, the reality for the state's aquatic ecosystems is likely much worse than this analysis shows. There is ample evidence that current water sampling procedures fail to capture the true maximum loads in Michigan surface waters (see Appendix B). It is also clear that EPA's benchmark levels are inadequate and that assessing neonics individually (when they are usually found as mixtures) ignores their clearly additive effects and possibly even synergistic impacts.

Current science shows that ecological degradation is real and occurring in real time as a result of neonic water contamination. Aquatic contamination by neonic insecticides has been a worldwide problem since their introduction (Morrissey et al. 2015), and Michigan is no exception. A full description of issues as well as the historical context behind the registration of neonics can be found in previous detailed reports, especially Mineau and Palmer 2013, and Mineau and Kern 2023.

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2. NEONIC USE IN MICHIGAN

Neonicotinoid pesticides, or "neonics," 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, and will not be considered any further in this report. Most of the concern surrounding neonics stems from the use of the first three chemicals on this list, in part because of their toxicity and persistence, but also because these have been the most widely used neonic active ingredients (a.i.) in North America. There are currently no registered thiacloprid products in the United States.

2.1. Agricultural uses of neonic active ingredients

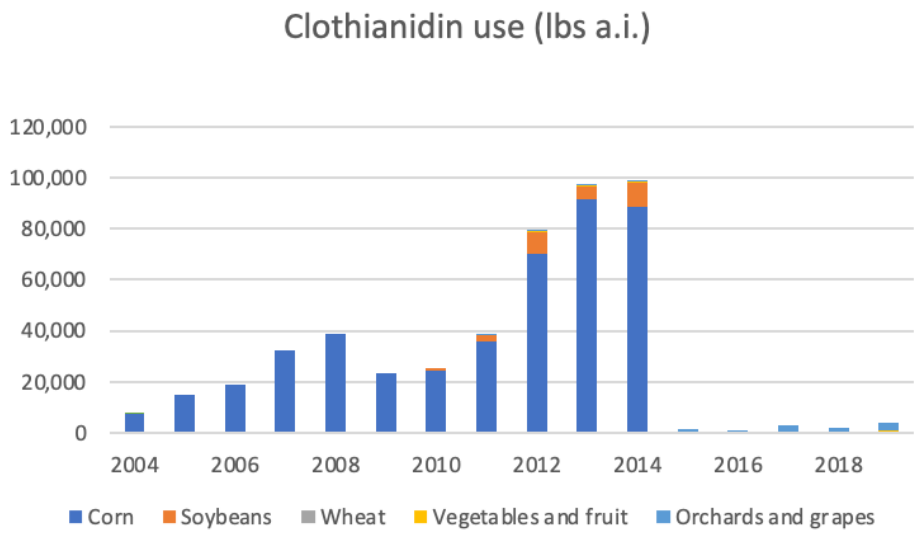
Researchers from the United States Geological Survey (USGS), Thelin and Stone (2013), developed a methodology for estimating agricultural pesticide use on both the state and county level through confidential surveys of pesticide application patterns across various crop types. These data, combined with crop acreage figures, allowed for extrapolation of use rates by region. Two estimates, E-Pest Low and E-Pest High, were produced. E-Pest High is particularly beneficial when pesticide use data are missing for a county, as it interpolates data from nearby counties based on crop area in order to avoid unrealistic zero estimates. These interpolated results are utilized in this report.

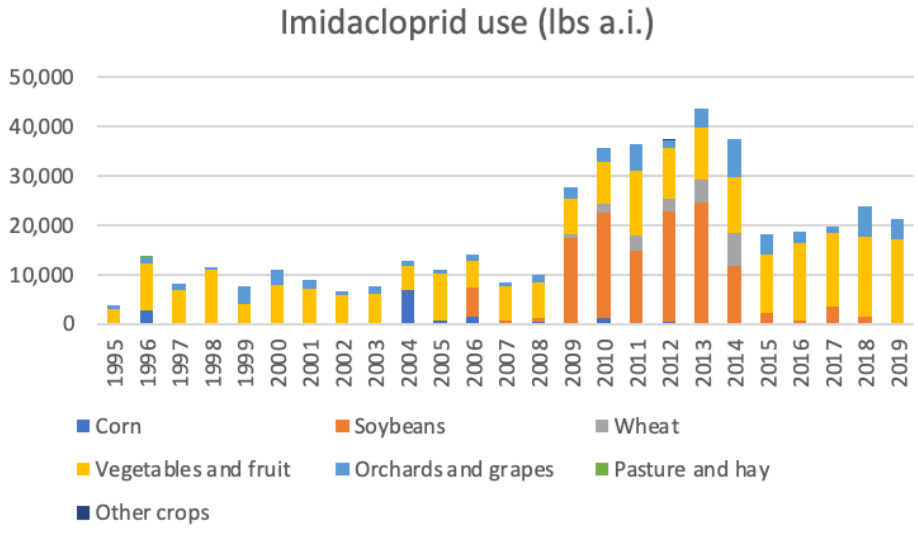
Continuing this work, Wieben (2021) released data spanning 1992 to 2019, noting that the 2018 and 2019 figures were preliminary.

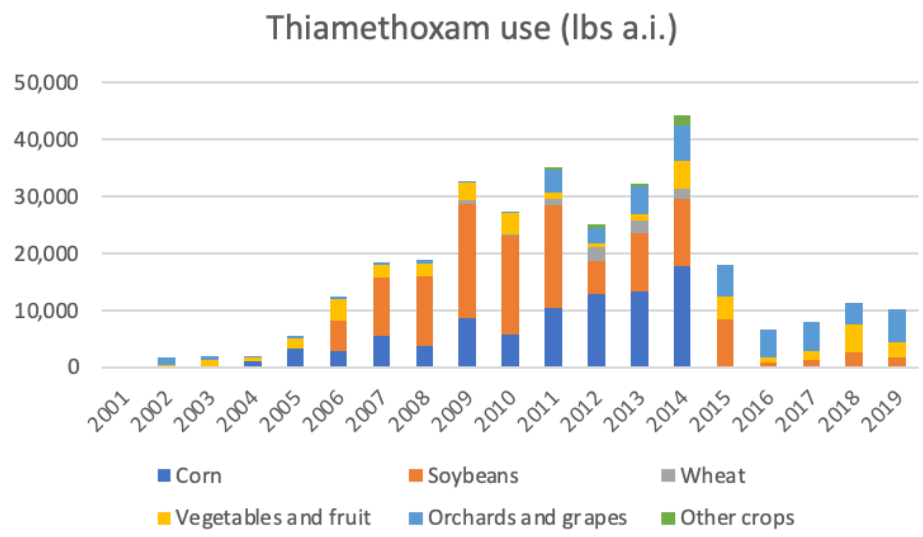
The data presented here (Figures 1a, b, c, d, and e) focus on high-acreage crops (corn, soybeans, wheat, cotton, and alfalfa) and aggregated low-acreage crops (e.g., vegetables and fruit, orchards and grapes, pasture and hay, and other crops) consolidated at the state level. These estimates reflect only agricultural pesticide use in Michigan, omitting domestic, landscape, and industrial applications. Not including these other uses means that we notably underestimate use of imidacloprid, which is extensively applied to turf and ornamental plants.

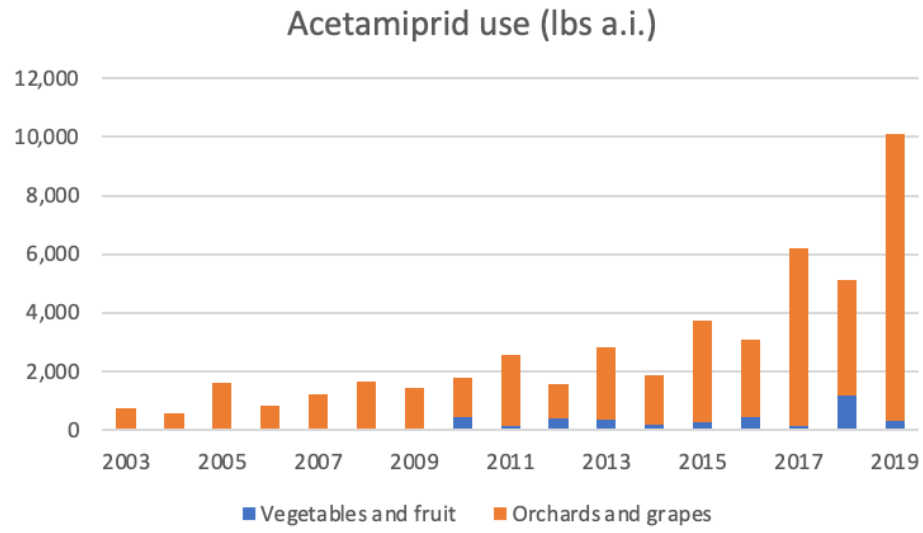
After 2014, data for neonic seed treatment use were no longer collected. The sudden drop in the estimated use of clothianidin, imidacloprid, and thiamethoxam between 2014 and 2015 reflects this change in data collection efforts and highlights the overwhelming share of the neonic market that seed treatments represent. The use of particular neonic chemicals within particular seed markets is also clear, with clothianidin dominating the corn market, imidacloprid being popular in the soybean market, and thiamethoxam split between corn and soybean.

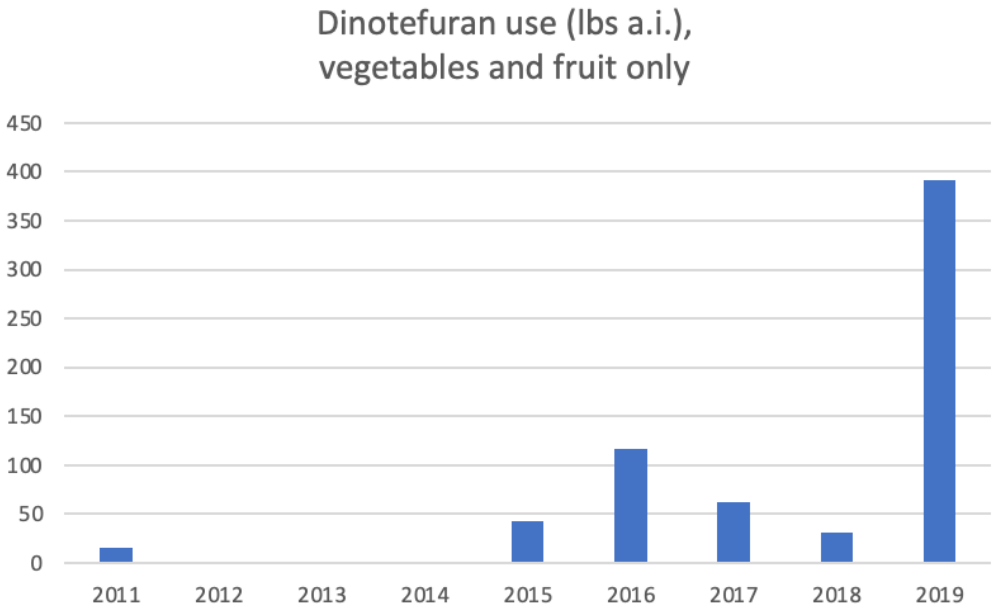
Figures 1a–e. USGS estimates (in pounds of active ingredients) of clothianidin (a), imidacloprid (b), thiamethoxam (c), acetamiprid (d), and dinotefuran (e) use in Michigan by main crop group for 2001 to 2019. Note that the Y axis differs from graph to graph; the relative length of the bars is therefore not meaningful. The abrupt declines in estimated use in 2015 for the main neonics clothianidin, imidacloprid, and thiamethoxam are not real but reflect the fact that seed treatments were not included after 2014. Clothianidin, especially, shows the most important drop, indicating that most of that compound is used as a seed treatment.





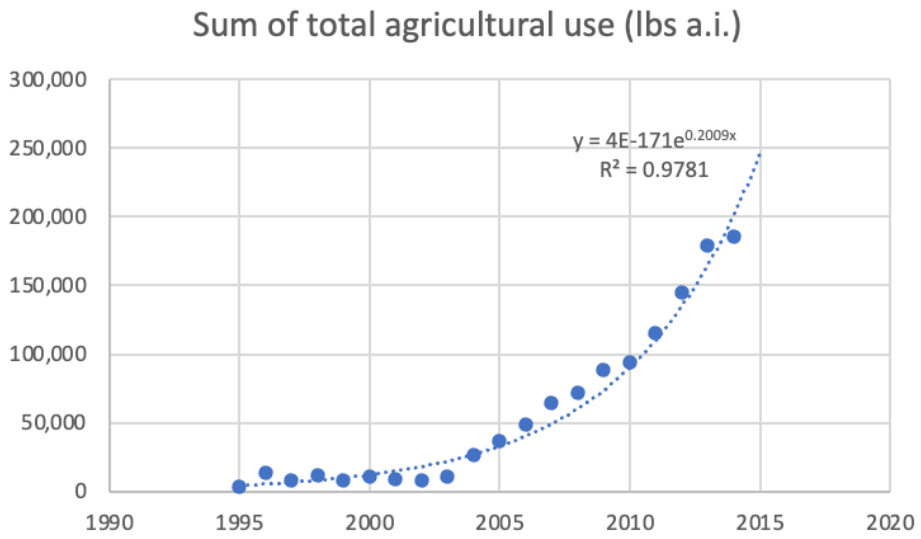

Thiamethoxam use (lbs a.i.)
50,000
40,000
30,000
20,000
10,000
0
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
Corn
Soybeans
Wheat
Vegetables and fruit
Orchards and grapes
Other crops


Acetamiprid use (lbs a.i.)
12,000
10,000
8,000
6,000
4,000
2,000
0
2003
2005
2007
2009
2011
2013
2015
2017
2019
Vegetables and fruit
Orchards and grapes


Dinotefuran use (lbs a.i.),
vegetables and fruit only
450
400
350
300
250
200
150
100
50
0
2011
2012
2013
2014
2015
2016
2017
2018
2019

Summing all the neonic uses, yearly use increased exponentially between first registration and 2014 (Figure 2). This pattern is not unique to Michigan; it is very typical of the neonic use patterns seen across North America.

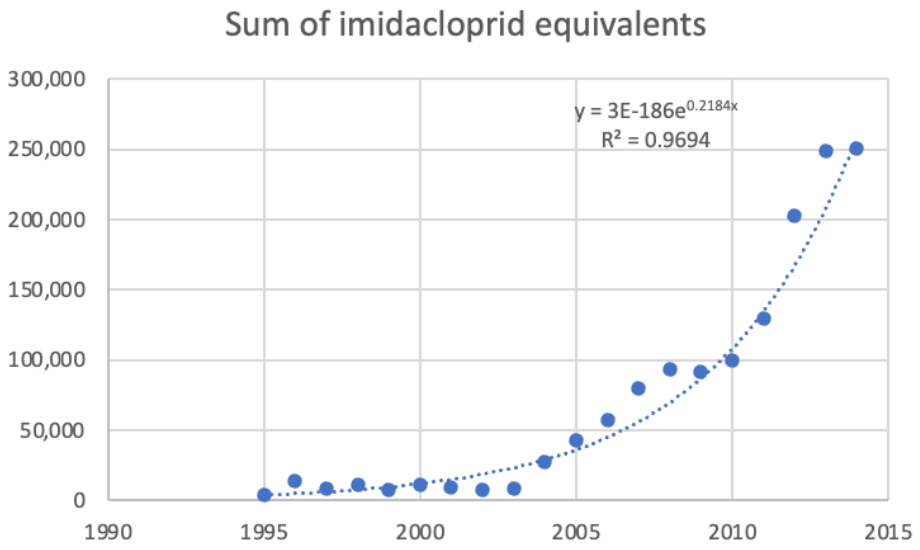
Figure 2. Sum (in pounds of active ingredients) of all neonic insecticides used in agriculture in Michigan. The R^2 value is a commonly used regression statistic that estimates how well the fitted exponential regression line explains the data. In this case, the curve explains almost 98% of the variance in the data, which is a very good fit.



We can estimate that, following the same relationship, the year 2015 should have recorded about 240,000 pounds a.i. of all neonics combined. Without tracking seed treatment use, the estimated total in 2015 was 42,000 pounds. This means that roughly 84% of the total agricultural neonic use in Michigan by weight was in the form of seed treatments: primarily clothianidin on corn; imidacloprid on soybeans, wheat, and corn; and thiamethoxam on soybeans and corn.

While the three main neonic active ingredients are not equally toxic to aquatic ecosystems, Appendix A shows how toxicity across the three chemicals can be assessed by calculating imidacloprid-equivalent toxicity through comparative toxicity tests conducted on the same species. This approach is the preferable one, given that water toxicity benchmarks are highly dependent on the extent of information available and this varies greatly among the different molecules (see Appendix A for a full discussion). The conclusion from the analysis outlined in Appendix A is that clothianidin is the most toxic of the main three neonics – almost twice as toxic as imidacloprid. Thiamethoxam is approximately half as toxic as imidacloprid, but this estimate is complicated by the fact that thiamethoxam breaks down in the environment and turns into clothianidin. Clearly, the largest potential threat to aquatic systems in Michigan has been the exponential increase in clothianidin. Figure 3 charts this total toxic load in Michigan, and the curve shows the same exponential increase from the first days of neonic registration in the mid 1990s to 2014, when seed treatments ceased to be included in surveys.

Figure 3. Calculated imidacloprid equivalents in pounds a.i. for the agricultural use of the main three neonics – imidacloprid, clothianidin, and thiamethoxam – in Michigan.



2.2. Non-agricultural use of the three main neonic active ingredients

Data on non-agricultural uses of neonics are not available for Michigan. However, data from other states with a pesticide accounting system (e.g., Minnesota; see Mineau 2024) show that, as the use of imidacloprid in corn and soybean was gradually replaced by clothianidin and thiamethoxam, there was an offsetting increase in imidacloprid use for structural, landscape, and veterinary uses. This gave rise to a greater degree of contamination from urban use, either through direct runoff or after passage through stormwater or wastewater treatment plants (WWTPs). Indeed, research shows that neonics are not removed, or are only partially removed, by treatment. For example, Xie et al. (2021), studying California WWTPs, estimated that 92% of the imidacloprid entering the WWTPs went through untouched.

Although non-agricultural uses still typically account for a small fraction of total use in corn- and soybean-producing states, they do expand the geographic areas where neonics are used and, as a result, where neonic water contamination is typically found. Use of imidacloprid in Michigan likely follows these national trends.

2.3. Increasing neonic use means increasing toxic potential for harm to aquatic ecosystems

Neonics have become ubiquitous contaminants of aquatic ecosystems worldwide, with demonstrated harms. A number of characteristics make them particularly problematic: They are highly persistent in soils; they are highly water soluble, migrating easily and often through runoff; they are very toxic to a broad range of species; and they are harmful at concentrations that are often too low to be detected. Their main use by volume is as prophylactic coatings applied to crop seeds before planting – seed treatments – which research increasingly shows provide little if any economic benefits to farmers under most conditions.[1]

1 A full agronomic review is beyond the scope of this report, but see US EPA (2014), Douglas and Tooker (2015), Douglas et al. (2015), Krupke et al. (2017), and 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; and 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.

As early as 1994, scientists at the U.S. Environmental Protection Agency (EPA) warned that both acute and chronic aquatic risk triggers had been exceeded for both non-endangered and endangered species exposed to imidacloprid (Mineau and Palmer 2013). A full 30 years later, these exceedances have come to pass and aquatic systems are being systematically degraded by neonic use in Michigan and elsewhere. A full description of issues as well as the historical context behind the registration of these neonics can be found in previous detailed reports – especially Mineau and Palmer 2013, and Mineau and Kern 2023.

3. EVIDENCE OF WATER CONTAMINATION IN MICHIGAN

Analyzing water samples for residues is the conventional way of assessing pesticides' potential environmental impacts, and this section summarizes the available data for Michigan. However, it is important for the reader to note that interpretation of the results is not straightforward and that water contamination levels cannot be taken at face value. Appendix B describes the main problems with the approach; some of those considerations appear in the sections below.

3.1. Core USGS water contamination data

Data were accessed through the Water Quality Portal maintained by USGS and EPA under the National Water Monitoring Council umbrella.[2] It combines the extensive USGS database and other water quality data collected by EPA for "states, tribes, watershed groups, other federal agencies, volunteer groups, and universities through the Water Quality Exchange framework." For Michigan, only USGS data were located. A total of 850 distinct reports of surface water analyses were inventoried for neonics and a few of their degradates between 2001 and 2023. This tally includes some samples where residue levels were recalculated by different methods, especially imidacloprid samples collected in 2015–2016 (see below). A large number of sampling sites consisted of only a single visit or very few visits, making non-detect results not probative of actual year-round concentrations.

Only imidacloprid was analyzed from 2001–2015, and at some sites thereafter, imidacloprid continued to be the only neonic active ingredient analyzed. Detection limits for imidacloprid were extremely variable and often elevated, frequently ranging from 25–100 ng/L, well above the currently published 10 ng/L EPA benchmark for harm to aquatic ecosystems (see full discussion of benchmarks in Appendix A). Failure to analyze for clothianidin and thiamethoxam at most of the USGS sites is unfortunate, as an accurate water contamination picture demands analyzing for all the major neonics, given prevailing trends in product substitution. Over time, imidacloprid seed treatments have been replaced by clothianidin and thiamethoxam treatments. Only an assessment of the combined residues, therefore, allows a true understanding of current neonic contamination patterns and their impact on the aquatic environment.

USGS sampled for imidacloprid at 77 locations. Detections occurred at 22 sites. However, the proportion of sites with detections (29%) is not very meaningful for a number of reasons. All but one of the sites with no detection were sampled only once, or at most twice, over the 24 years of sampling. There is a direct relationship between the frequency of sampling and the probability of detection as well as the peak levels detected (e.g., see Mineau 2024 for Minnesota). Even when there is more frequent sampling, the proportion of positive samples typically reported is likewise not meaningful without context – for example, without knowing whether samples originated from an area where pesticides were used, or whether they came from agricultural, urban, or mixed watersheds. In addition, samples are taken at various times of the year and from various types of water bodies, including large rivers where the dilution factor is very high. Impacts to aquatic life are expected where most of the aquatic productivity is taking place – in small drainage ditches and ponds bordering field areas to small feeder streams. Neonic contamination, when detected, is often at higher levels in these areas.

Only 50 analyses of clothianidin and thiamethoxam were recorded in total, with 48 of these at sites with some neonic detection. All these analyses were in the context of a yearlong sampling program (2015–2016) at four monitoring sites on the St. Joseph, Grand, Saginaw, and Rouge Rivers (see section 3.2). The high proportion of clothianidin at the first three

2 https://www.waterqualitydata.us/. Accessed 8 April 2024.

of these sites suggests that, as predicted from use data, clothianidin may be a much more important neonic contaminant in the state, despite the fact that the maximum clothianidin detections exceeded the maximum imidacloprid detections at only one of the four sites.

The question we should be asking is whether neonics, where used, are being seen at concentrations that cause harm to the receiving environments. It is therefore more meaningful to look at the data from those sites where a neonic was detected at least once during the 2001–2023 time interval, indicating some use in the watershed. There is clear evidence from the literature that, where neonics are used on crops, they will be detected in nearby bodies of water at a very high frequency. Reported values for sites with at least one detection between 2001 and 2023 are summarized in Table 1 below. The proportion of positive detections as well as the maximum values reported are given.

Table 1. A summary of USGS surface water analyses for Michigan of the three main neonic active ingredients between 2001 and 2023. The number of samples taken, the proportion above the detection level, and the maximum concentration detected are given. All sites with at least one neonic detection are included. CLO = clothianidin, IMI = imidacloprid, THI = thiamethoxam.

USGS Site	Body of water	CLO No. samples	CLO % detected	CLO Max detected value (ng/L)	IMI No. samples	IMI % detected	IMI Max detected value (ng/L)	THI No. samples	THI % detected	THI Max detected value (ng/L)
4101500*	St. Joseph River	12	50.00%	20.6	24	4.17%	3.4	12	8.33%	3
4116000	Grand River				1	100.00%	47.4			
4116004	Grand River				1	100.00%	49.1			
4118564	Grand River				1	100.00%	50.2			
4119065	Plaster Creek				1	100.00%	89.2			
41190654	Grand River				1	100.00%	53.3			
4119400*	Grand River	12	50.00%	7.9	25	76.00%	71.9	12	0.00%	
4157005	Saginaw River	12	100.00%	11.7	24	33.33%	36.5	12	25.00%	9.6
4157225	unknown				21	9.52%	15.6			
4157226	unknown				23	52.17%	49.4			
41572264	Tributary to Randall Drain				19	5.26%	27			
41572269	Tributary to Randall Drain				17	29.41%	2650			
4157227	unknown				26	38.46%	117			
4159064	Pinnebog River				3	33.33%	9.56			

Table 1. A summary of USGS surface water analyses for Michigan of the three main neonic active ingredients between 2001 and 2023. The number of samples taken, the proportion above the detection level, and the maximum concentration detected are given. All sites with at least one neonic detection are included. CLO = clothianidin, IMI = imidacloprid, THI = thiamethoxam.

USGS Site	Body of water	CLO No. samples	CLO % detected	CLO Max detected value (ng/L)	IMI No. samples	IMI % detected	IMI Max detected value (ng/L)	THI No. samples	THI % detected	THI Max detected value (ng/L)
4161820	Clinton River				267	59.18%	985			
4165500	Clinton River				12	83.33%	281			
4166500	River Rouge	12	16.67%	3.8	24	79.17%	340	12	0.00%	
414804086444001	Galien River				1	100.00%	78.6			
414900086413701	Galien River				1	100.00%	44.1			
415205086344401	Galien River				1	100.00%	20.4			
425626085272901	Grand River				1	100.00%	47.9			
425712085411001	Grand River				1	100.00%	53.7			

* Acetamiprid also detected at the site.[3]

With the exception of one site, maximum imidacloprid concentrations detected were all above the EPA benchmark threshold, where damage to the aquatic environment is expected.

3.2. Yearlong USGS sampling for the three main neonics in four major Michigan rivers

As mentioned above, USGS only tested for all three major neonic chemicals as part of a special project at four major river sites. This occurred between October 2015 and September 2016, and these sites are detailed in Table 2.

Table 2. Four Michigan USGS sites with yearlong sampling (2015–2016) for the three major neonics.

USGS ID number	Site description	Latitude	Longitude	Upstream land use based on satellite photographs
04101500	St. Joseph River at Niles	41.8292138	−86.2597325	Mixed, primarily urban
04119400	Grand River near Eastmanville	43.0241884	−86.0264354	Mixed, primarily agricultural
04157005	Saginaw River at Holland Avenue at Saginaw	43.4219699	−83.951918	Urban, refuge lands
04166500	River Rouge at Detroit	42.3730923	−83.2546513	Urban, parkland, golf course

3 The two acetamiprid detections were recorded in 2016, at 7 and 12 ng/L.

In particular, the St. Joseph, Grand, and Saginaw are major rivers with large flows and, therefore, high dilution factors. Ecological impacts are much more likely in small drains and tributaries. Only the Grand River site can be said to drain agricultural areas, at least proximally. Table 3 gives a snapshot of the presence of the three neonics for the 2015–2016 period over a whole year. Imidacloprid results were recalculated by USGS by alternate methods (DQCALC or Estimated Detection Levels) that have proved to be more robust and often yielded results two to three times higher than EPA's Method Detection Limits. However, for the sake of this comparison and for consistency with other samples, the values calculated with EPA's Method Detection Limits were retained.[4] To address the issue of multiple neonic residues per site, imidacloprid equivalents are calculated for all samples as per the method described in Appendix A.

Table 3. Intensive sampling results for the main three neonic insecticides in four Michigan rivers. Blank cells indicate that concentrations were below detection limits.

Sampling date	Location	Clothianidin (ng/L)	Imidacloprid (ng/L)	Thiamethoxam (ng/L)	Imidacloprid equivalents (ng/L)
2015-10-13	St. Joseph River				
2015-11-03	St. Joseph River				
2015-12-01	St. Joseph River				
2016-01-20	St. Joseph River	3.7			7.0
2016-02-02	St. Joseph River				
2016-03-08	St. Joseph River	6.4			12.2
2016-04-12	St. Joseph River	7.9			15.0
2016-05-03	St. Joseph River	13.8		3	27.8
2016-06-07	St. Joseph River				
2016-07-12	St. Joseph River	3.3			6.3
2016-08-09	St. Joseph River				
2016-09-13	St. Joseph River	20.6	3.4		42.5
2015-10-14	Grand River		5.2		5.2
2015-11-04	Grand River		4.4		4.4
2015-12-02	Grand River		3.4		3.4
2016-01-21	Grand River	6.6	8		20.5
2016-02-03	Grand River	3	5.1		10.8
2016-03-09	Grand River	3.2			6.1
2016-04-13	Grand River	6.4	4		16.2
2016-05-04	Grand River	7.9	7.4		22.4
2016-06-08	Grand River		5.3		5.3
2016-07-13	Grand River		17.4		17.4
2016-08-10	Grand River		7.4		7.4
2016-09-14	Grand River	6.1	8.2		19.8
2015-10-07	Saginaw River	2.5	2.6		7.4
2015-11-12	Saginaw River	2.9			5.5
2015-12-10	Saginaw River	2.3			4.4

4 Because values using the DQCALC procedure were often two to three times higher than the Method Detection Limit, Table 3 represents a conservative estimate.

Table 3. Intensive sampling results for the main three neonic insecticides in four Michigan rivers. Blank cells indicate that concentrations were below detection limits.

Sampling date	Location	Clothianidin (ng/L)	Imidacloprid (ng/L)	Thiamethoxam (ng/L)	Imidacloprid equivalents (ng/L)
2016-01-20	Saginaw River	5.1			9.7
2016-02-10	Saginaw River	7.5	2	3.7	18.2
2016-03-07	Saginaw River	4.8			9.1
2016-04-05	Saginaw River	11.7	3.3		25.5
2016-05-11	Saginaw River	9.1		3.3	19.0
2016-06-15	Saginaw River	5.3	4.7		14.8
2016-07-20	Saginaw River	5.1	8		17.7
2016-08-03	Saginaw River	11.1	13.8	9.6	40.0
2016-09-21	Saginaw River	3.6	6.1		12.9
2015-10-06	River Rouge		5.3		5.3
2015-11-09	River Rouge		5.2		5.2
2015-12-08	River Rouge				
2016-01-05	River Rouge		2.5		2.5
2016-02-09	River Rouge		5.5		5.5
2016-03-02	River Rouge		8.5		8.5
2016-04-04	River Rouge	3.8	7.6		14.8
2016-05-10	River Rouge	3.7	8.3		15.3
2016-06-14	River Rouge		21.8		21.8
2016-07-19	River Rouge		97.9		97.9
2016-08-02	River Rouge		153		153.0
2016-09-20	River Rouge		17.6		17.6

Clearly, even such relatively frequent sampling is inadequate to find true maxima, as explained in Appendix B. The Grand River, because of its more proximal agricultural drainage area, was expected to yield more consistent detections of clothianidin and thiamethoxam, the two neonics replacing imidacloprid on the coated seed market. Surprisingly, it was the Saginaw River that had more consistent detections of these two neonics. The Grand and Saginaw Rivers show evidence of yearlong contamination by both imidacloprid and clothianidin. As these samples were taken a decade ago, and neonic use is believed to have increased since then, the situation today is likely to be much worse.

River Rouge registered the highest contamination level with an imidacloprid measurement of 153 ng/L in August. Given its location, non-agricultural uses of imidacloprid are likely responsible for this elevated contamination level.

Based on imidacloprid equivalents, the vast majority of samples exceeded both the new European chronic standard estimated as 5.7–6.5 ng/L and the EPA published 10 ng/L benchmark. Although contamination levels tended to be higher in late summer and fall, they were consistently above the chronic impact level for the entire sampling interval, from October to the following September. Given the size of these rivers and their vast drainage areas, this "evening out" of residues is perhaps not surprising. It does, however, speak to the very broad contamination of the state's surface freshwater, given the size of the watersheds.

The presence of yearlong contamination is the reason chronic benchmarks are the appropriate ones to compare water concentrations to. Also, the cumulative toxicity potential of neonics has been well studied. In an expansion of previous analyses, Sánchez-Bayo and Tennekes (2020) showed the ability of many small neonic exposures to add up, causing greater and greater harm over time. They showed quite convincingly that neonics inflict irreversible cumulative toxicity in both aquatic and terrestrial invertebrates. This argues for using peak concentrations at any one site rather than time-weighted averages, as is often done when comparing concentrations to chronic benchmarks.

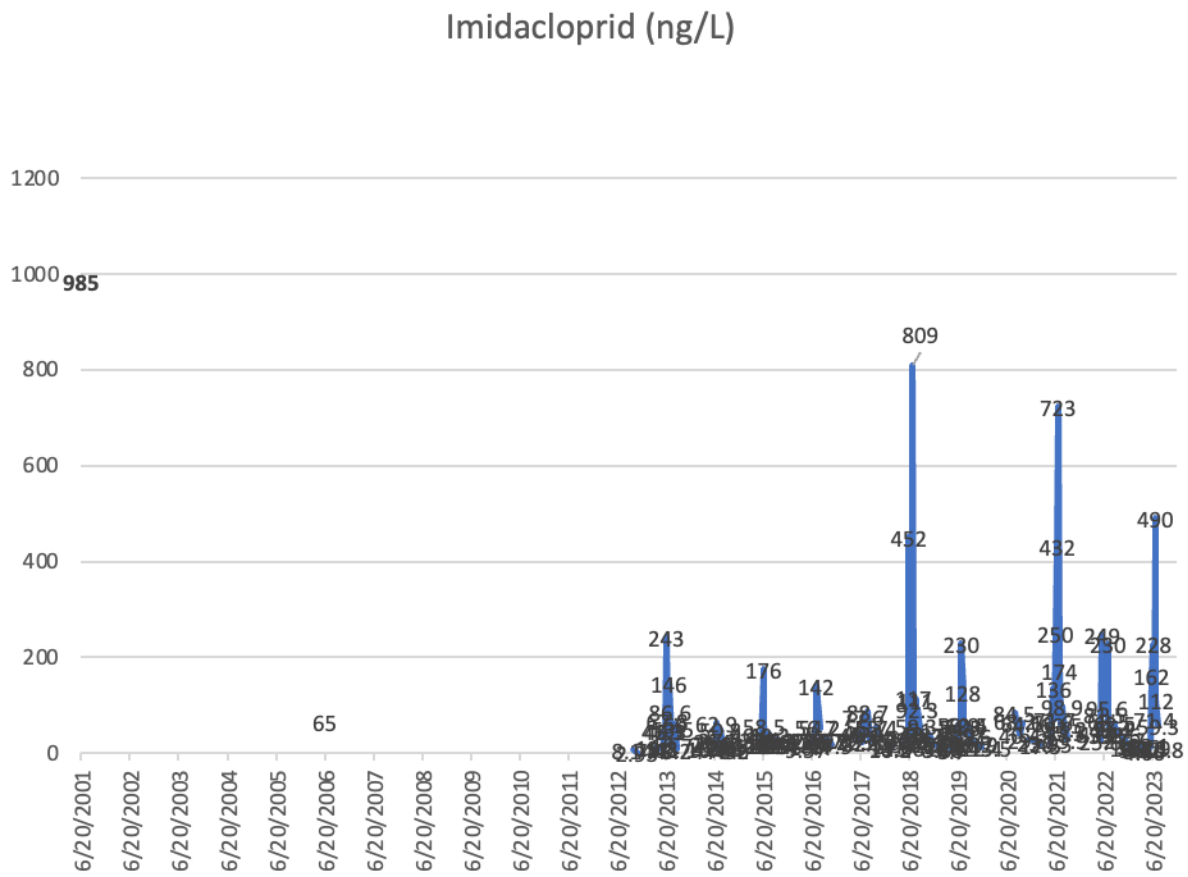
3.3. Long-term monitoring at specific USGS sampling sites

Despite being restricted to imidacloprid, the core USGS data offer results of longer-term monitoring at a few sites. The record for the highest level of imidacloprid detected in Michigan belongs to site USGS-041572269, a tributary of the Randall Drain near Akron that had a reading of 2,650 ng/L in 2018. That site is clearly embedded into a highly agricultural area and is likely a reflection of near-field conditions (Figure 4). Other neonics were not measured at this site but would almost certainly have been present given the predominance of clothianidin and thiamethoxam seed treatments; this would add greatly to the overall toxic load of the drain to Saginaw Bay.

Figure 4. Satellite map (courtesy of Google Earth) of location of site USGS-041572269, one of the Randall Drain sampling sites. The yellow pin marks the location. The body of water to the northwest is Saginaw Bay.

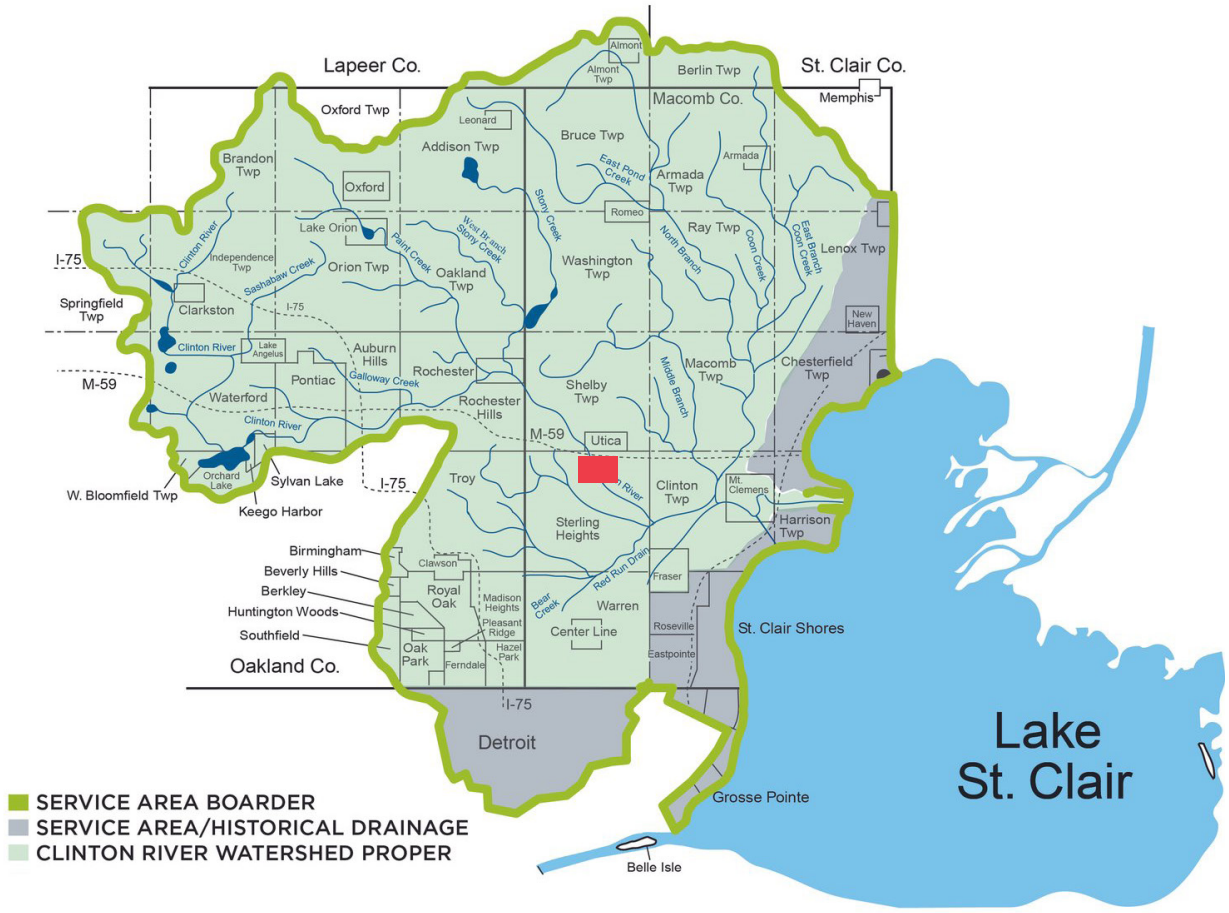
Fully 39% of all neonicotinoid reported analyses for Michigan were carried out at a single sampling point, USGS 04161820 on the Clinton River at Sterling Heights. Unfortunately, imidacloprid was the only neonic analyzed at this site, but the site nonetheless offers the best temporal view of how streams and creeks receive repeated contamination above injury levels. This station has some of the highest recorded levels of imidacloprid (viz. recorded maxima of 985 ng/L in 2001, 809 ng/L in 2018, and 723 ng/L in 2021), second only to the Randall Drain mentioned earlier (Figure 5).

Figure 5. Imidacloprid analyses for the Clinton River at Sterling Heights from 2001 to 2023.



Very few samples were taken from 2001 to 2012; nevertheless, it is easy to see that peak levels were often above 100 ng/L and therefore expected to have substantial impacts on the aquatic environments, even before the presence of other neonics is considered. The apparent decline of imidacloprid peak values between 2001 and 2023 (if real) is expected to be more than compensated by increases in clothianidin – more prone to runoff and more toxic to aquatic life. The map reproduced in Figure 6 shows the location of the Clinton River watershed.

Figure 6. Map of the Clinton River watershed (courtesy of the Clinton River Watershed Council). The red rectangle shows the approximate location of the USGS monitoring site detailed in Figure 7.

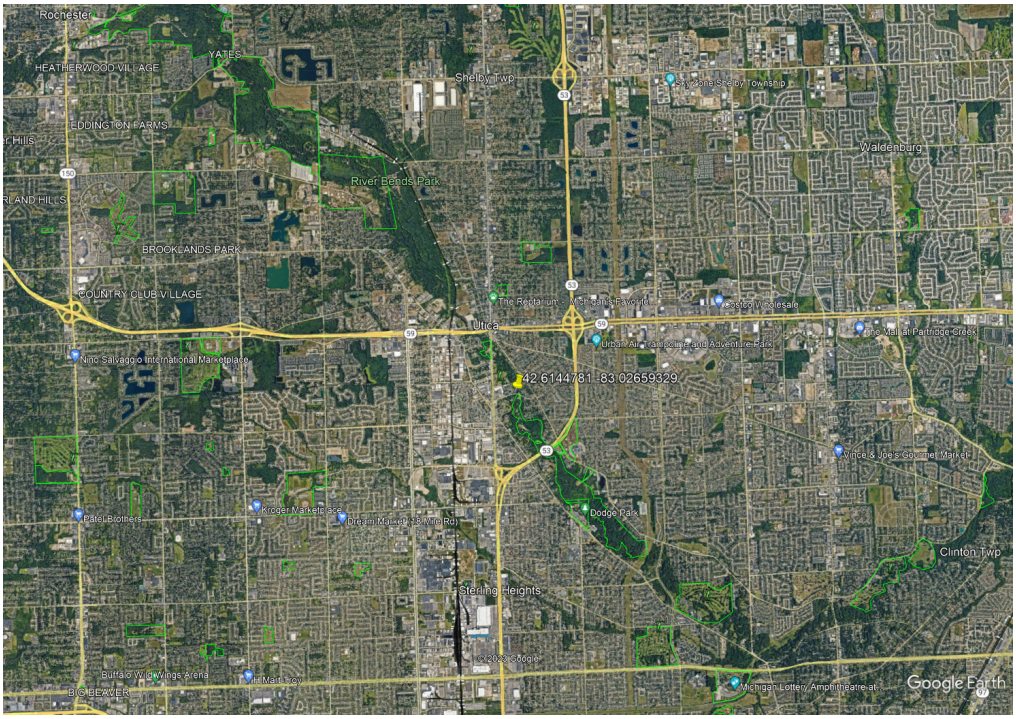


The Clinton River runs for 81.5 miles to Lake St. Clair. The Clinton River Watershed Authority describes the river as *"a valuable freshwater resource that not only provide[s] important ecological functions but also provide[s] the region with many important uses, including water-oriented towns, tourism, diverse wildlife habitat, boating, fishing, and many other recreational activities."*[5] The satellite image below (Figure 7) clearly shows the effort to protect and buffer the river through parks and other green spaces. Unfortunately, the repeated contamination from imidacloprid at such high concentration means that aquatic life is chronically impacted, even before the other neonics or any other contaminants or stressors are considered. In fact, as detailed in Appendix B (Section B.3.1), the available data underestimate the real maxima – likely by a factor of 10 or more. This is clearly an issue with imidacloprid, which is already being measured at levels 50 to 100 times higher than the chronic benchmark.

Upstream of the sampling station, one of the tributaries to the Clinton River is Paint Creek, the only designated cold-water trout stream in the state. Unfortunately, although there is a USGS water sampling station on Paint Creek, no tests for neonic residues were conducted there.

5 https://www.crwc.org/clinton-river-watershed#:~:text=The%20Clinton%20River%2C%20its%20watershed,and%20many%20other%20recreational%20activities. Accessed January 2025

Figure 7. Satellite map (courtesy of Google Earth) of site USGS 04161820 on the Clinton River at Sterling Heights. The yellow pin marks the location.



3.4. Water contamination data from Michigan's Department of Agriculture and Rural Development and Department of Environment, Great Lakes, and Energy

In 2023, Michigan's Department of Agriculture and Rural Development (MDARD) collaborated with the Department of Environment, Great Lakes, and Energy (EGLE) to test for neonics in state surface waters.[6] Six neonics – acetamiprid, clothianidin, dinotefuran, imidacloprid, thiacloprid, and thiamethoxam – were analyzed, along with a few breakdown products. Detection levels were generally quite low, most often ranging between 3.1 and 6.2 ng/L. A total of 54 creeks and rivers were sampled, most of them four times during the year.[7] Neonics were detected at exactly half of the sites. The bulk of detections were of the three neonic active ingredients used as seed treatments – imidacloprid, clothianidin, and thiamethoxam. Acetamiprid was detected at one site only and at a low level; dinotefuran was detected at two sites only, with no other neonics detected at those sites.

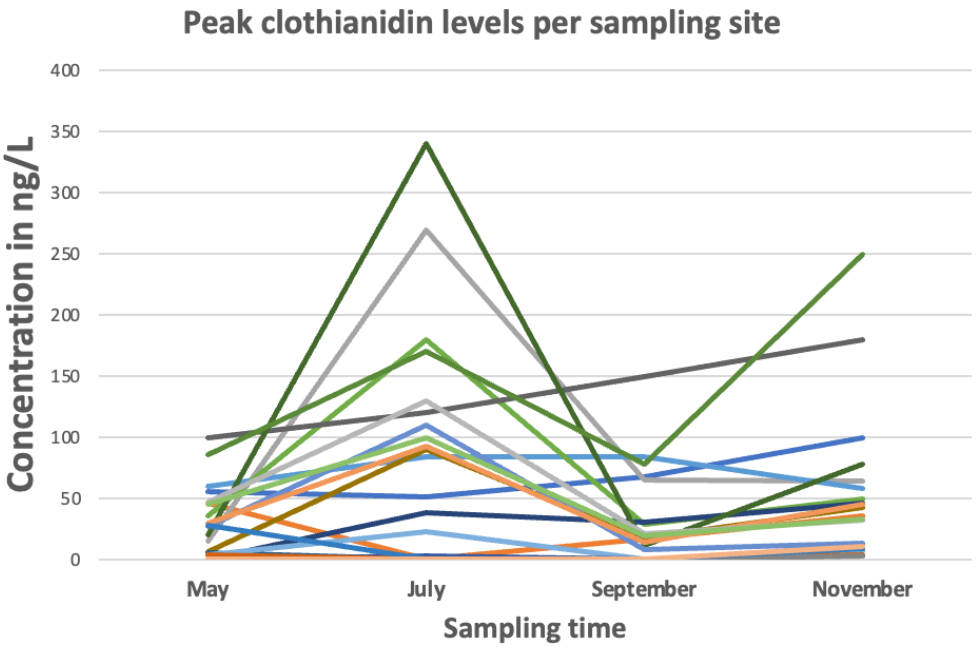
Sampling dates were May 1–24, July 5–19, September 6–27, and November 1–13. According to Michigan State University Extension Service,[8] the optimal time for planting corn is early to mid May; for soy, late April to mid May is optimal, although more factors such as pest pressure and seeding density enter into the decision regarding planting date. This suggests that, depending on the presence of rainfall, the impact of spring seeding on the extent of neonic contamination may not have been captured by the May sampling dates. Peak residues are expected following the first rains after use – seeding in the case of seed treatments. Indeed, the data suggest that spring peaks associated with seeding were missed as a result of the sampling dates chosen (Figure 8). Looking at clothianidin, the compound most associated with seed treatment use, it is clear that levels in July are routinely higher than May levels. The data also show a frequent increase in November, likely associated with increased fall precipitation and runoff.

6 I am grateful to EGLE for sharing these data.

7 One river (Pine River) was sampled at two different locations; another labeled as a tributary to Peterson Lake was sampled only once.

8 https://www.canr.msu.edu/news/what_is_the_best_time_to_plant_corn_in_michigan; https://www.canr.msu.edu/news/soybean-planting-and-time-management-considerations. Accessed November 2024.

Figure 8. Clothianidin levels at river and creek sites with positive detections.



The need to consider all neonics – and not merely imidacloprid, as was the case in most of the limited USGS sampling (see section 3.1) – is clear from the MDARD-EGLE data. Two or more neonics were present at 75% of the sites with a neonic detection. Appendix A shows the difficulty of choosing a credible impact benchmark to compare water concentrations to, especially with the less well-studied neonics such as clothianidin and thiamethoxam. For that reason, imidacloprid-equivalent concentrations were calculated as described in Appendix A for each sampling site and date; the result can be compared directly with the various imidacloprid benchmarks such as the more scientifically defensible European Union (EU) acute benchmark of 62 ng/L, or EPA's 10 ng/L currently published chronic benchmark, or the more protective EU chronic benchmark of 6.25 ng/L.[9]

Assessing the full impact of the observed residues is difficult, knowing that peak levels were likely missed in most cases. Nevertheless, one-time peak levels of the sampled neonics are given in Table 4 along with the one-time maximum value of imidacloprid equivalents summed for each sampling period. Sites with no neonic detections are listed in Table 5. All sampling locations are mapped in Figure 9 along with the presence of various crops and whether they are field crops often associated with a seed treatment.[10] Although one should not infer too much from a single year of sampling (and should keep in mind the sampling time constraints referred to earlier), the association between neonic detection and the presence of field crops likely to receive a seed treatment is clear. The fit is not perfect, and as seen elsewhere (e.g., Mineau 2024 for Minnesota), sites in predominantly non-crop areas can also show the presence of neonics. Because of the characteristics of neonics, a single user can contaminate a creek or stream to the point where biological impacts are expected.

9 Appendix A reviews benchmark setting in detail and discusses why EPA's methodology is not scientifically defensible, resulting in an acute benchmark that is wildly divergent from current scientific thinking. The currently published EPA acute benchmark for imidacloprid is 385 ng/L; the EU has set its acute benchmark at 62 ng/L (the average of two scientifically defensible estimates: 57 ng/L and 68 ng/L). Also, it is clear that, given the demonstrated season-long contamination at sites, chronic benchmarks should be prioritized when trying to understand the damage to aquatic environments.

10 Map prepared by Maeve Sneddon based on U.S. Department of Agriculture crop data accessed November 21, 2024, at https://croplandcros.scinet.usda.gov/. Note that some crops in the group "other crops" (e.g., potatoes) are also likely to receive seed treatments.

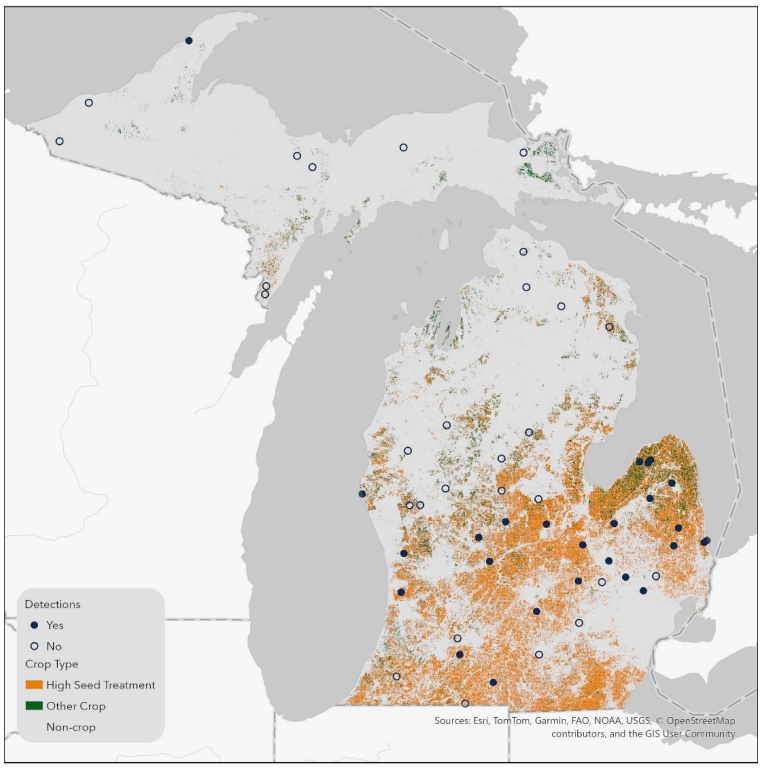
Table 4. Summary of MDARD and EGLE neonic sampling in 2023 for sites where there were neonic detections. The highest of the four yearly values is given for each active ingredient as well as the imidacloprid equivalents for the three main neonics as described in Appendix A.

Sampling site	Latitude	Longitude	Maximum acetamiprid (ng/L)	Maximum clothianidin (ng/L)	Maximum dinotefuran (ng/L)	Maximum imidacloprid (ng/L)	Maximum thiamethoxam (ng/L)	Maximum IMIDACLOPRID EQUIVALENTS for three main neonics (ng/L)
Bad River	43.3071	−84.3688	0	100	0	0	7.3	193.9
Begunn Creek	47.2472	−88.5155	0	0	8.2	0	0	0.0
Birch Creek	43.16921	−82.50776	0	46	0	0	25	100.7
Burtch Creek	43.15204	−82.54243	0	270	0	0	84	557.5
Dickerson Creek	43.1934	−85.1557	0	7.6	0	0	0	14.4
Elkton Drain	43.81941	−83.19086	0	84	0	0	98	211.5
Fletcher Drain	43.27274	−82.83706	0	180	0	730	19	1072.0
Johnson Drain	42.1945	−85.3752	0	46	0	0	0	87.4
Kanouse Lake Drain	42.82381	−84.00071	0	7.9	0	0	0	15.0
Kearsley Creek	42.85874	−83.45144	0	0	43	0	0	0.0
Macatawa River (South Branch)	42.7298	−86.05486	0	180	0	9.7	22	363.4
Mill Creek	43.1241	−82.8926	0	90	0	11	16	190.5
Millington Creek	43.311	−83.583	0	11	0	0	0	20.9
Misteguay Creek	43.13125	−83.94888	0	340	0	75	150	800.5
Mud Creek	43.52189	−83.16789	0	110	0	0	59	240.3
North Branch Mill Creek	43.12402	−82.89623	0	93	0	13	19	199.8
Pigeon River	43.82943	−83.2883	0	130	0	3.2	250	379.5
Pine River at Hillis Rd.	43.32517	−84.84424	0	0	0	0	4.3	2.3
Pine River at Porter Rd.	43.5171	−84.4621	0	23	0	0	2.5	43.7
Pinnebog River	43.84428	−83.16207	0	100	0	4.6	54	223.2
Prairie Creek	42.99154	−85.02868	0	4.2	0	0	0	8.0
South Fork Cass River	43.65303	−82.91658	0	4.3	0	0	0	8.2
Stony Creek	43.5594	−86.5069	0.55	3	0	0	0	5.7
Thread Creek	42.9969	−83.6478	0	0	0	11	0	11.0
Trib to Cold Creek	41.9563	−84.9858	0	28	0	0	0	53.2
Trib to Crockery Creek	43.05896	−86.02386	0	250	0	0	4.8	477.5
Trout Creek	42.74111	−83.24601	0	0	0	29	0	29.0
Willow Creek	42.5662	−84.4821	0	11	0	0	0	20.9

Table 5. List and location of sites sampled by MDARD and EGLE in 2023 where no neonics were detected above detection levels (typically 3.1–6.2 ng/L).

Sampling site	Latitude	Longitude
Adams Drain	42.815	-83.72548
Betts Creek	43.6057	-85.5409
Black River	46.44981	-90.01707
Canada Creek	45.11577	-84.1976
Chippewa River	43.58809	-84.88839
E B Chocolay River	46.33292	-87.26168
East Pond Creek	42.8658	-83.0984
Fawn River	41.7746	-85.3107
Fox River	46.4002	-86.0275
Gull Creek	42.33495	-85.40178
Honey Creek	42.4675	-83.9909
Kelley Creek	45.28064	-87.62038
Little River	45.2139	-87.6331
Lost Creek	46.75637	-89.678
McMahen Creek	46.359	-84.6347
Mullet Creek	45.56	-84.6365
North Branch Cedar River	44.07533	-84.57174
North Branch Pine River	44.13452	-85.52586
Overton Creek (aka Littlefield Creek)	43.8567	-84.8907
Pere Marquette River	43.92225	-85.97699
Trib between Kimball Lk and Ryerson Lk	43.46725	-85.83378
Trib to Fremont Lake	43.46401	-85.95715
Trib to Peterson Lake	42.19477	-84.4557
Unnamed Tributary to Dowagiac River	42.00768	-86.108099
West Branch Sturgeon River	45.2716	-84.6021
West Branch Whitefish Creek	46.2431	-87.0838
Wolf Creek	44.946	-83.6393
Pine River at Porter Rd.	43.5171	-84.4621

Figure 9. Map of MDARD and EGLE 2023 neonic sampling sites, showing whether neonics were detected at the site and crop information. Areas labeled as potential high seed treatment areas are those where field crops such as corn, soy, cereals, and canola are grown.



Even though sampling missed the time of year with the highest probability of detecting peak levels, peak imidacloprid equivalents were above the EPA chronic benchmark at 77% of the sites where it was possible to calculate it. In fact, the acute benchmark recently developed by the EU[11] was exceeded at more than half of the sites where neonics were detected, at times by more than tenfold. This suggests neonics inflict significant and widespread damage to aquatic life in Michigan.

This more recent snapshot of the current situation in Michigan also shows that clothianidin dominates most of the samples and accounts for most of the toxic impacts, highlighting the inadequacy of federal sampling that has not included clothianidin residues in its core sampling program. Clothianidin dominated surface water samples also in Minnesota (Mineau 2024), which has much more comprehensive sampling by the state department of agriculture. Additionally, Hladik et al. (2014) showed the dominance of clothianidin in Midwest samples associated with corn and soybean production.

4. ACKNOWLEDGMENTS

The author would like to thank Daniel Raichel, Maeve Sneddon, Rachel Garwin, Lucas Rhoads, and Lena Freij of the Natural Resources Defense Council (NRDC) for their contributions and input into this report and NRDC for providing the financial support to produce it. The views expressed in this report remain those of the author. This report comes close on the heels of a sister report (Mineau 2024) that analyzed data for the state of Minnesota. Readers will find some commonality between these two reports, especially in the appendixes.

11 See Appendix A for a discussion of how the 61 ng/L benchmark recently developed by the EU for imidacloprid is much more scientifically defensible than the current EPA equivalent.

APPENDIX A: SETTING TOXICITY BENCHMARKS FOR NEONIC INSECTICIDES

A.1. EPA's water quality benchmarks: Background and shortcomings of EPA's methodology

To assess a chemical's potential impact on aquatic systems, it is essential to estimate the concentration in water at which adverse effects on aquatic life are expected, referred to as a "benchmark" value. This value is typically set by gathering toxicity test data – ideally from a diverse range of organisms – and extrapolating from these data to derive a single metric that protects the aquatic ecosystem as a whole. Different jurisdictions often derive different benchmark values for the same chemicals due to varying approaches.

EPA (U.S. Environmental Protection Agency) has traditionally used a single test value for what it terms the "most sensitive" species, i.e., the lowest acute or chronic toxicity value among those available. This approach implies comprehensive protection for all species but can be misleading due to its dependence on often limited testing. Even closely related species can exhibit significant differences in sensitivity to pesticides or other chemicals. The likelihood of identifying the "most sensitive" species is much higher if many species are tested rather than just a few. However, datasets for newer pesticides are typically too small (sometimes comprising only one or two species) to reliably identify the true "most sensitive" species in ecosystems that contain thousands. Consequently, even where contamination levels are maintained below such a benchmark, aquatic systems can, and often do, suffer damage.

Recognizing these issues, most other jurisdictions or regulatory bodies have adopted alternative strategies. One approach involves placing all available toxicity endpoints (e.g., LC_{50} values – the concentration expected to kill half of the tested organisms) on a mathematical distribution and selecting a single value based on the proportion of values expected to fall below this chosen point. The 5% tail of a distribution is often used as the benchmark, although sometimes the 10% or 15% tail is selected. In addition, this tail value can be estimated with a high (e.g., 95%) or low (e.g., 50%) probability of not being overestimated and leaving several species without the needed protection. Methods have been developed to approximate the results of a distribution analysis when there are too few values to plot a distribution. An alternate strategy is to acknowledge that the "most sensitive" species cannot logically be determined and that even distribution analyses have limitations, particularly with small sample sizes. Thus, an extrapolation or safety factor (either arbitrarily derived or more frequently based on experience with similar datasets) is applied to the lowest value found in the tested species sample or to a value derived by curve-fitting as described above. Using a safety factor also accounts for the possibility that wild organisms may be more sensitive than laboratory test organisms for various reasons. Of all these approaches, EPA's is the least protective and the least scientifically defensible.

A comprehensive examination of the process for setting reference levels for imidacloprid and other neonics was detailed in a series of reports focusing on New York State (Mineau 2019) and California (Mineau 2020). These reports argued that EPA had systematically underestimated the toxicity of clothianidin and thiamethoxam (and other, lesser-known neonics), as it had initially done with imidacloprid. A significant finding was that, as of 2017, with 36 aquatic invertebrate species tested, sensitivity to imidacloprid varied by a factor of 790,000 from the least to the most sensitive aquatic insect or crustacean. Thus, setting any benchmark based on a "most sensitive" species from smaller datasets on other neonics is as scientifically rigorous as a roll of the dice, even after EPA applies a factor of 2 to the lowest recorded test value.[12] EPA does use species sensitivity distributions in some cases, although its approach is not consistent. This has led to inconsistencies in the benchmarks and misguided views as to which neonic is the most toxic to aquatic life. In contrast, the EU has a much more scientific approach to deriving aquatic toxicity benchmarks (European Commission 2018). Its approach uses species sensitivity distributions where possible, although assessment factors are still used on the results to reflect the quantity and quality of available data.

12 https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk.

As an example, Tables A.1 and A.2 summarize the long and checkered history of acute and chronic freshwater benchmarks for imidacloprid derived by regulatory agencies and scientists on the basis of data available to them at the time. This is to illustrate the difficult and arbitrary nature of setting protective benchmarks, even for a pesticide as intensively studied as imidacloprid.

Table A.1. A historical summary of acute imidacloprid benchmarks for freshwater invertebrates.

Source	Benchmark (ng/L)	Justification
USEPA (1994)	18,700	Based on the mysid shrimp – the lowest of freshwater and saltwater species multiplied by LOC (level of concern) of 0.5.
USEPA (2007)	34,500	Lowest of three tests examined – to which a factor of 2 has been applied in keeping with the 0.5 LOC for a risk quotient.
EFSA (2008)	550	European Food Safety Authority, the EU regulatory authority for pesticides. Lower of two species tested to which a factor of 100 has been applied in keeping with Annex VI triggers for the Toxicity/Exposure Ratio.
RIVM (2008) (Netherlands – nonregulatory)	200	Maximum acceptable concentration from short-term exposure or exposure peaks and threefold safety factor.
Nagai et al. 2012	430	HC513 from a species sensitivity distribution (SSD) methodology, which combines species within the same genus – predicted with 50% confidence.
USEPA (2012)	35,000	Aquatic life benchmark online – accessed by Mineau and Palmer 2013 – presumably the same methodology as regulatory review.
Mineau and Palmer (2013)	1,010	HC5 (with 50% confidence) for acute exposure in crustacea.
Mineau and Palmer (2013)	1,020	HC5 (with 50% confidence) for acute exposure in insects.
Mineau and Palmer (2013)	220	HC5 (with 50% confidence) for acute exposure in all aquatic invertebrates (ignoring lack of normality).
EFSA (2014) European regulatory	98	Median estimate of the HC5 of 490 ng/L based on all insect studies (N=15) divided by safety factor of 5. Incidentally, the lower 95% bound of the HC5 was also determined to be 98 ng/L.
Morrissey et al. (2015)	200	Lower confidence interval of HC5 from SSDs generated using 138 acute toxicity (LC50) and 37 chronic toxicity (LC/EC50) tests considering all neonicotinoid compounds and all species. Intended to be applied to summed residues of all neonicotinoids.
PMRA (2016)	360	Acute HC5 for 32 species tested.
Bayer Crop Science (2016) (from EPA 2016)	1,730	HC5 after removal of several studies; rejected by USEPA 2017 because of biased acceptance of data points.
USEPA (2016)	385	Based on quantitatively acceptable mayfly study from open literature and factor of 2.
PMRA (2021)	540	Revised analysis (from 2016) based on re-selection of available studies following industry comments.
USEPA (2022)	1,430	Analysis in the context of endangered species assessment with revised endpoint selection and deletion of numerous studies, an industry approach initially rejected in 2016. See details in text below.

13 Stands for Hazardous Concentration at the 5% tail of a distribution of concentration values – here for a distribution of LD_{50} values.

Table A.1. A historical summary of acute imidacloprid benchmarks for freshwater invertebrates.

Source	Benchmark (ng/L)	Justification
SCHEER (2021) European Union (Science Advisory)	65	New analysis by SCHEER (Scientific Committee on Health, Environmental and Emerging Risks) using a deterministic approach.
SCHEER (2021) European Union	57	New analysis by SCHEER using a probabilistic approach.

Table A.2. A historical summary of chronic imidacloprid benchmarks for freshwater invertebrates.

Source	Benchmark (ng/L)	Justification
USEPA (1994)	160	Lowest NOAEC of FW and SW species – mysid shrimp.
USEPA (2007)	1,000	Obtained with an acute/chronic ratio. (Using the usual chronic NOAEC for Daphnia would have meant accepting a value of 800,000 – much higher than the acute value).
CCME (2007) (Canada – nonregulatory)	230	EC15 for the most sensitive of two freshwater species tested chronically to which a factor of 10 has been applied.
EFSA (2008)	200	European Food Safety Authority. NOAEC (600 ng/L) from a 21-day German microcosm study to which an assessment factor of 3 has been applied based on expert deliberations.
Dutch Regulatory Authority (2008) from RIVM 2008	13	Maximum permissible concentration (MPC) for Dutch ecosystems.
RIVM (2008) (Netherlands – nonregulatory)	67	Maximum permissible concentration for long-term exposure derived from lowest NOAEC value and assessment factor of 10. This replaces the older value of 13 ng/L above.
USEPA (2012)	1,050	Aquatic life benchmark online – accessed by Mineau and Palmer 2013 – methodology uncertain.
Mineau and Palmer (2013)	29	Distribution analysis of NOAECs for chronic studies on seven single species and one species assemblage.
Mineau and Palmer (2013)	8.6	Second proposed method. The higher of two empirically determined acute-chronic ratios for insects applied to the most sensitive insect species of the eight tested to date.
RIVM (2014)	8.3	Updated maximum permissible concentration (MPC) for long-term exposure derived from chronic studies NOAEC/LC_{10}/EC_{10} using SSD approach and HC_5 with assessment factor of 3 applied.
Vijver and Van den Brink (2014)	30	Proposed as relevant threshold based on chronic EC_{10} for two mayfly species after the work of Roessink and colleagues.
EFSA (2014)	9.0	Chronic HC5 of 27 ng/L based on 10 studies from the literature. The assessment was based on the Netherlands analysis of the data. Experts agreed to apply a safety factor of 3.
Morrissey et al. (2015)	35	Lower confidence interval of HC_5 from SSDs generated using 37 chronic toxicity tests considering all neonicotinoid compounds and all species. Intended to be applied to summed residues of all neonicotinoids.
Smit et al. (2015)	170	Following a review of five mesocosm studies. However, see comment about underrepresentation of sensitive species.
PMRA (2016) (Canada regulatory)	41	Pest Management Regulatory Agency. Chronic HC_5 for 10 species.

Table A.2. A historical summary of chronic imidacloprid benchmarks for freshwater invertebrates.

Source	Benchmark (ng/L)	Justification
Bayer Crop Science 2016 – as Moore et al. (2016)	1,010	HC_5 from a selection of microcosm and mesocosm studies. Selection process criticized by PMRA and European Food Safety Authority.
USEPA (2016)	10	NOAEC (No Adverse Effect Concentration) for mayfly study from open literature.
PMRA (2021)	160	Revised approach using higher-tier mesocosm data; this approach had been criticized earlier by the PMRA as having limitations on the number of species tested. The more "traditional" approach based on a probabilistic assessment of chronic studies yielded a value of 11 ng/L.
SCHEER (2021) European Union (Science Advisory)	2.4	New analysis by SCHEER (Scientific Committee on Health, Environmental and Emerging Risks) using a deterministic approach.
SCHEER (2021)	6.8	New analysis by SCHEER using a probabilistic approach.
Schmidt et al. (2022)	17	Published distribution analysis of new chronic results combined with existing values from the literature.

It is clear that, as time progressed, EU regulators grew increasingly concerned about the aquatic impacts of imidacloprid contamination – but not so their North American counterparts. Most chronic benchmarks developed in the EU have hovered around 10 ng/L or lower for a number of years now; it has most recently been set as low as 2.4 ng/L under the European Union's water framework initiative. The published EPA 10 ng/L benchmark is reasonable in this context. It is in line with current thinking by many experts worldwide and appears to fit the current field evidence. In their regulatory function, however, both EPA and the Canadian Pest Management Regulatory Agency (PMRA) have become less stringent and are now recommending higher benchmarks and reduced protection for aquatic systems.

It is sobering to realize the significance of a 2.4 ng/L benchmark – the one developed by SCHEER in 2021 – when most of the analyses presented in this report had detection limits of 25 ng/L, 10 times higher.

A.1.1. EPA's neonic aquatic life benchmarks are currently far less protective than those established in the EU.

Aside from perhaps the currently listed imidacloprid chronic benchmark of 10 ng/L, EPA benchmarks for the main neonic insecticides are still out of step with those of European regulatory agencies and, as argued in the text on methodological grounds, not sufficiently protective.

As Table A.3. below demonstrates, the benchmarks developed by EU regulators are all more protective than those used by EPA, often by more than an order of magnitude for the main three neonics. The case of the Canadian PMRA is a bit stranger. The latter, following an extensive review of aquatic toxicity of the three main neonics between 2016 and 2018, had benchmarks more closely aligned with those of European regulators. However, following the publication of its proposed decision to cancel many registrations because of aquatic concerns, and after consideration of industry comments, chronic benchmarks were radically increased – by approximately tenfold in the case of clothianidin and thiamethoxam. The PMRA no longer proposes to de-register any of the three main neonics.

Table A.3. Comparison of USEPA aquatic freshwater benchmarks[a] with those in Canada and the EU.

Active ingredient	Acute (ng/L)			Chronic (ng/L)		
	USEPA acute benchmark[a]	PMRA online benchmark[b]	EU published benchmark[g]	USEPA online benchmark[a]	PMRA online benchmark	EU published benchmark
Imidacloprid	385	540[c]	57–65[h]	10	160[d]	5.7–6.8[h]
Thiamethoxam	17,500	9,000	550–770[i]	740	300[e]	43[i]
Clothianidin	11,000	1,500	340[j]	50	120[f]	10[j]
Thiacloprid	18,900	20,400	80[k]	970	680	10[k]
Acetamiprid	10,500	12,000	160[l]	2,100	5,000,000	37[l]
Dinotefuran	>484,150,000	Reference to EPA benchmark	Not determined	>95,300,000	Reference to EPA benchmark	254[m]

a Data obtained from: https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk (consulted November 2024; said to have been updated October 22, 2024). However, these do not reflect more recent assessments such as the USEPA 2022 endangered species assessments (see text).

b Available online from: https://www.canada.ca/en/health-canada/services/consumer-product-safety/pesticides-pest-management/public/protecting-your-health-environment/programs-initiatives/water-monitoring-pesticides/aquatic-life-reference-values.html (consulted November 2024; said to have been updated May 15, 2024).

c Based on PMRA (2021); revised from 360 ng/L in PMRA (2016).

d Based on PMRA (2021); revised from 41 ng/L in PMRA (2016).

e Revised from 26 ng/L (based on a distributional analysis) in PMRA (2018a).

f Revised from 1.5 ng/L (based on a distributional analysis) in PMRA (2018b).

g The range in values reflects the use of different methodologies – deterministic versus probabilistic.

h SCHEER 2021.

i SCHEER 2023a.

j SCHEER 2023b.

k SCHEER 2023c.

l SCHEER 2023d.

m EU (European Union) 2014. Dinotefuran is registered as a biocide in the EU. Although not a formal benchmark, the European Union has set this PNEC (predicted no effect concentration) for risk assessment purposes.

None of the benchmarks for any of the agencies, however, address the issue of multiple neonic residues at sampling sites, which is commonplace in the United States as it is in many other jurisdictions. To assess the real risk of aquatic impacts and to avoid issues regarding smaller datasets (applicable to all the neonics but imidacloprid), I believe a comparative approach is more fruitful (as discussed below).

A.1.2. EPA unjustifiably discarded its own imidacloprid aquatic life benchmarks in its recent endangered species assessments

Recent developments in the U.S. risk assessment world merit a short discussion. By 2016, EPA had finally adopted benchmarks for imidacloprid that were more in line with those of other regulatory bodies in Europe and Canada – namely 10 ng/L. However, under the guise of standardizing and improving data quality, recent assessments by EPA on the toxicity of neonics to threatened and endangered aquatic life unjustifiably used less protective risk assessment benchmarks (USEPA 2022).[14] This shift abandoned the measurable harms or "endpoints" that EPA previously had relied on to assess pesticide threat to aquatic species, and also intentionally excluded a number of studies identifying the harms of neonics to species at exceptionally low concentrations.

In its previous assessment of imidacloprid, EPA (USEPA 2016) supported the use of "immobilization" – i.e., the pesticide concentration at which organisms were paralyzed and rendered nonfunctional from an ecological standpoint – as the appropriate endpoint. It stated that "*the effects of imidacloprid (and other neonicotinoids) on mayfly immobilization occur at*

14 Risk assessment endpoints and benchmarks established by EPA for harm to aquatic life clearly serve two different purposes. However, one may question the logic of using risk assessment endpoints that are radically less protective than established benchmarks of protection, especially when the assessment is for an endangered species.

substantially lower levels than lethality. Specifically, LC_{50} [lethality] *values ranged from 6.7 to 154* µg *ai/L for* C. dipterum *and* C. horaria *whereas* EC_{50} [immobilization] *values varied from 0.77 to 32* µg *ai/L for these same species.*"

It also stated 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)

This assessment method is consistent with that of most aquatic toxicologists. Yet, in its most recent assessment of the three principal neonics as they relate to threatened and endangered species – i.e., those most vulnerable to extinction – EPA inexplicably adopted new data exclusion principles and altered how study endpoints are evaluated to make its assessments less protective. Despite previously emphasizing the ecological importance of immobilization in laboratory tests, the agency favored mortality endpoints over immobilization: "*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).*"

It also imposed stricter conditions on studies, leading to the exclusion of many independent university research studies. 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 2022, Appendix 2-5, p. 3)

In this revised assessment, EPA did not reference any of its previous assessments or explain the rationale behind rejecting immobilization as a critical endpoint or dismissing test data based on formulated material as opposed to technical-grade material. This shift means that industry tests now hold more weight in toxicity assessments, as independent researchers often lack access to technical-grade material.

Moreover, while EPA (USEPA 2022) claims that its data selection process introduces more scientific rigor in deriving benchmarks, it commits a serious methodological error by including multiple data points for the same species, thereby skewing the distribution. For example, the cladoceran species *Daphnia magna*, which is known to be highly insensitive to neonics, is included six separate times in the distribution analysis (USEPA 2022, Appendix 2-5).

For its 2016 acute toxicity standard for imidacloprid (USEPA 2016), EPA had utilized immobilization values from three ephemeroptera species, ranging from 650 to 1,400 ng/L. The 650 ng/L value (from Alexander et al. 2007) was deemed "qualitative" due to the lack of raw data, while the 770 ng/L value from Roessink et al. (2013) was adopted as the freshwater acute standard. These tests were conducted with formulated materials (typical end-use products, or TEP). The acute toxicity benchmark of 385 ng/L was derived by applying a safety factor of 2 to this "quantitatively acceptable" mayfly endpoint, acknowledging the likelihood of more sensitive, yet untested, species. In the same assessment (USEPA 2016), a species sensitivity distribution of 32 acute values produced an HC_5[15] of 360 ng/L. The close agreement between these two values likely reassured EPA scientists, who then used the 385 ng/L acute benchmark, which still appears on the agency's website. Additionally, in the same report, EPA dismissed an attempt by Bayer Crop Science (cited as Moore et al. 2016) to establish an HC_5 value at 1,730,000 ng/L, citing clear bias in the selection of acceptable data points.

Yet, without justification, EPA adopted a significantly less protective acute imidacloprid benchmark in line with the industry proposal rejected earlier – 1,430 ng/L (1,100 ng/L for insect species) and 13,150 ng/L for freshwater and saltwater invertebrate species, respectively – for its assessment of mortality to threatened and endangered aquatic or consumer species. This change was based on HD_5 values after a distribution analysis of carefully selected data, substituting mortality for immobilization where possible, and excluding tests with formulated material or insufficient dose levels. A sublethal maximum acceptable toxicant concentration (MATC) of 280 ng/L based on the most acceptable chronic study is also used in the risk calculations (USEPA 2022, Appendix 4-2). This revisionism contrasts starkly with the European Food Safety Authority's assessment of imidacloprid, which revised its acute benchmark downward, pegging it at 57 or 65 ng/L depending on the method followed. If this new EPA interpretation were to stand and replace the current EPA published benchmark, this would represent a more than fourfold difference in what is considered a safe concentration in Europe versus the United States, based on the prevention of sublethal impacts in the case of endangered species, or more than a twentyfold difference for lethal effects on individuals of unlisted species.

While EPA appears to continue to endorse its 2016 385 ng/L benchmark on its website, the benchmark is effectively meaningless if it is discarded any time actual protection or mitigation is required. It is ironic that this reversal and

15 Hazardous concentration (in this case LC_{50} value) at the 5% tail of the fitted distribution, as explained earlier.

effective "downgrading" of the toxicity of imidacloprid occurs in the context of an assessment intended to protect endangered species.

A.2. Toward more defensible aquatic toxicity benchmarks: How do neonics compare in their aquatic toxicity?

In our prior report (Mineau and Palmer, 2013), we advocated that the aquatic toxicity of thiamethoxam and clothianidin to aquatic insects and crustacea should be regarded as akin to that of imidacloprid, based on comparisons of toxicity tests conducted on the same species with different neonics. This assertion was reaffirmed and bolstered by Morrissey et al. (2015), who concluded: "*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 scholars have also remarked on the comparable toxicity of imidacloprid and second-generation neonics like clothianidin and thiamethoxam, e.g., Hoyle and Code (2016), leveraging newer data such as that of Cavallaro et al. (2017, but accepted for publication and data made available in 2016). The latter obtained comparative data for the three neonics on the same chironomid species, revealing nearly identical toxicities for imidacloprid and clothianidin and slightly less for thiamethoxam.

Publication of additional comparative data by Raby et al. (2018a, 2018b) finally furnished enough information to convince EPA that differences among neonic active ingredients were indeed minimal (USEPA, 2020a): "*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 (LC_{50} values are 2x and 5x higher than imidacloprid; confidence bounds do not overlap with those of imidacloprid or clothianidin).*" (USEPA, 2020a)

Similar findings emerged from chronic toxicity tests, with thiamethoxam being marginally less toxic than imidacloprid, albeit by only a twofold difference. This is reflected in current EU benchmarks (see Table A.3). It is noteworthy that thiamethoxam breaks down into clothianidin, thus diminishing the ecological relevance of its lesser toxicity. No-effect concentrations[16] for clothianidin and imidacloprid were within a factor of 4 and 2 for the most sensitive and second-most sensitive species, respectively. Clothianidin proved more toxic than imidacloprid to the most sensitive species, a mayfly, but less toxic than imidacloprid for the second-most sensitive species, a chironomid. Maloney et al. (2018b) found that under simulated field conditions, chironomid populations were equally affected by imidacloprid and clothianidin, while thiamethoxam appeared to be about one-tenth as toxic.

Evidently, the differential toxicity attributed to the three main neonic active ingredients in past and present EPA aquatic risk assessments lacks scientific justification. Indeed, EPA has contradictory views on the relative toxicity of neonics. While EPA recognizes that, when fairly compared, their toxicity to aquatic life is similar (at least for the three main compounds), the official aquatic benchmarks are still very far apart. **For example, the clothianidin acute benchmark is 45-times less protective than that of imidacloprid when, in fact, clothianidin is nearly twice as toxic when tested on the same assemblage of organisms.** Whether imidacloprid or second-generation clothianidin demonstrates greater across-the-board toxicity also depends on whether acute or chronic values are considered (see Table A.4), further underscoring the inadequacy of the EPA methodology and the agency's disparate benchmarks for the chemicals.

At a minimum, neonics should be deemed of equivalent toxicity until proved otherwise (but see analysis below). This includes dinotefuran, the neonic active ingredient for which we have the least data. Given that water samples typically contain several neonic residues, an additive model of effect serves as a pragmatic starting point for evaluating the genuine impacts of neonics. However, we can propose better than a straightforward addition of residues with the three principal neonics for which more aquatic data have accumulated. I believe there are now sufficient data to work out toxicity equivalency factors for imidacloprid, clothianidin, and thiamethoxam.

16 This is the level in a toxicology study at which the endpoint being sought, e.g. lethality, is not seen. The no-effect level is highly dependent on the sample size used in the test as well as on the specific test conditions. It is less reliable than a computed LC_{50}, for example.

The EU recently gathered available aquatic toxicity data on the three main neonics and calculated distribution-based endpoints (e.g., HC_5 values calculated with a high confidence that they have not been overestimated) for both acute and chronic tests (SCHEER 2021, 2023a, 2023b). These are the same datasets that formed the basis of the new EU benchmarks (Table A.4).

Table A.4. Results of distribution analysis for aquatic invertebrates (SCHEER 2021, 2023a, 2023b)

Compound	Acute HC_5 ng/L	95% CL	Chronic HC_5 ng/L	95% CL
Clothianidin	336[a]	17.7–1,876	10.8	0.136–115.9
Imidacloprid	259	46–910	27.4	2.99–120
Thiamethoxam	7,721	1,587–23,760	Not provided	

a This value was not retained by EU authorities because of poor distributional fit and wide confidence limits.

I used those vetted compilations of toxicity tests assembled for all crustacean and aquatic insect tests (SCHEER 2021, 2023a, 2023b) to fairly compare the toxicity of the main three neonics to the same species (Table A.5). Most of the comparative tests were conducted in the same laboratory and therefore provide the best information on relative toxicity.

Acute toxicity tests were used for this analysis, both because there are more available comparisons and because chronic test conditions are more likely to diverge over time. Data were matched for test conditions, and only studies with the highest reliability ratings assigned by the EU were used. Test results were standardized by assigning a value of 1 to imidacloprid results. EC_{50} (immobilization) and LC_{50} results are compared separately because these are often generated from the same studies and would not be independent. The vast majority of the comparisons are from the same laboratory, most from Raby et al. (2018a, 2018b), mentioned earlier.

Table A.5. Comparison of acute crustacean and insect tests on the main neonics. Statistics derived for relative LC_{50} values, imidacloprid being set as 1.

Criterion of relative toxicity	EC_{50} for clothianidin	EC_{50} for thiamethoxam	LC_{50} for clothianidin	LC_{50} for thiamethoxam
No. of compared species	13	15	16	17
Range of relative toxicity endpoints	0.03–1.7	0.24–45	0.014–8.6	0.11–69
Arithmetic mean	0.76	7.8	1.75	9.8
Geometric mean	0.53	3.0	0.76	2.4
Median	0.52	1.9	0.82	2.2
% of species with equal or higher sensitivity	69%	27%	63%	29%

I would argue that the medians of the EC_{50} ratios provide the best starting point for establishing toxicity equivalents when adding up residues in any one sample. EC_{50} refers to paralysis or immobility of the test organism; this is easier to measure than mortality in some organisms. Also, it is the ecologically relevant measure in terms of ensuring a functioning aquatic ecosystem as argued by EPA in 2016 (but not in its 2022 assessment for endangered species), especially in a river system where affected individuals will be swept downstream if paralyzed (invertebrate drift). As the mean values are clearly influenced by a few extreme values, when the mean ratio is considered, clothianidin jumps from being nearly twice as toxic as imidacloprid to being a little under half as toxic. I posit that those same extreme values are responsible for the different probabilistic-based analyses, and that the median value provides the best insight as to the true relative

ecological toxicity of these two chemicals. At the end of the day, a higher proportion of the tested species (69% based on EC_{50} values) are more sensitive to clothianidin than to imidacloprid.

On that basis, toxicity equivalency factors of 1.9 for clothianidin (reciprocal of 0.52) and 0.53 for thiamethoxam (reciprocal of 1.9) are indicated. This means that clothianidin is roughly twice as toxic as imidacloprid, while thiamethoxam is roughly half as toxic. This differential is also consistent with the spread between their relative chronic toxicities. Again, the wide differential in current EPA benchmarks is not warranted and is indicative of poor methodology compounded by unequal datasets.

It is important to note that thiamethoxam is a proto-neonic and that much of its insecticidal activity comes from the fact that, after it is applied, thiamethoxam converts to clothianidin in the environment. In terrestrial environments, the yield of clothianidin from thiamethoxam is about 66% (European Commission 2006). It is not clear from the literature what the conversion of thiamethoxam to clothianidin in the external and internal environments of exposed aquatic invertebrates is likely to be. Therefore, the factors proposed here will be used to provide toxicity values in imidacloprid equivalents, recognizing that the impact of a mixture containing thiamethoxam is likely greater than calculated because it readily converts to clothianidin in the real world.

The full list of comparable data is given in Table A.6. Values with a high degree of reliability (1 or 2 in the EU scheme) were retained. When there were repeat measurements for the same endpoints under the matching conditions, a geometric mean of the values was computed.

Table A.6. Comparison of individual species tests for the three main neonics. Toxicities in ug/L.

Species	Clothianidin	Imidacloprid	Thiamethoxam	Matching conditions
Aedes sp.	29	41	67.4	Mortality, 48h, active substance, same study
Americamysis bahia	53	59	4,100	LC_{50}, 96h, active substance, different studies
Americamysis bahia	48	92	4,100	EC_{50}, 96h, active substance, different studies
Asellus aquaticus		84	78	EC_{50}, 48–96h, active substance, different studies
Asellus aquaticus		20,000	2,300	LC_{50}, 48–96h, active substance, different studies
Caecidotea sp.	537	321	4,775	EC_{50}, 96h, active substance, same study
Caenis sp.	122		382	LC_{50}, 96h, active substance, same study
Cheumatopsyche sp.	1,281	325	170	LC_{50}, 96h, active substance, same study
Cheumatopsyche sp.		176	119	EC_{50}, 96h, active substance, same study
Chironomus dilutus	3.4	2.5	36.8	EC_{50}, 96h, active substance, same study
Chironomus dilutus	12	12	61.9	LC_{50}, 96h, active substance, same study
Chironomus dilutus	5.93	4.63	55	LC_{50}, 96h, active substance, same study
Chironomus riparius	29		48	EC_{50}, 48h, active substance, different studies
Cloeon sp.	3,940	1,152	4,634	LC_{50}, 96h, active substance, same study
Cloeon sp.		23	44	EC_{50}, 96h, active substance, same study
Coenagrion sp.	14,556	3,463	15,061	LC_{50}, 96h, active substance, same study
Crangon uritai	260	570	820	EC_{50}, 96h, active substance, same study
Crangon uritai	360	2,200	2,200	LC_{50}, 96h, active substance, same study
Ephemerella sp.	19	11		EC_{50}, 96h, active substance, same study
Ephemerella sp.	587	68	335	LC_{50}, 96h, active substance, same study
Gammarus pulex	56.6	110		EC_{50}, 48h, active substance, different studies
Gyrinus sp.	41	58	14	EC_{50}, 96h, active substance, same study

Table A.6. Comparison of individual species tests for the three main neonics. Toxicities in ug/L.

Species	Clothianidin	Imidacloprid	Thiamethoxam	Matching conditions
Gyrinus sp.	63	132	31	LC_{50}, 96h, active substance, same study
Hexagenia sp.	5.5		35.8	EC_{50}, 96h, active substance, same study
Hyalella azteca	4.8	177	391	EC_{50}, 96h, active substance, same study
Hyalella azteca	5.2	363	801	LC_{50}, 96h, active substance, same study
McCaffertium sp.	1,328	1,810		LC_{50}, 96h, active substance, same study
McCaffertium sp.		10.6	81.7	EC_{50}, 96h, active substance, same study
Micrasema sp.		15	32.8	LC_{50}, 96h, active substance, same study
Neocleon triangulifer	3.5	5.2	5.5	LC_{50}, 96h, active substance, same study
Neocleon triangulifer	3.5	3.1	5.5	EC_{50}, 96h, active substance, same study
Nitocra spinipes	6.9	25	120	EC_{50}, 96h, active substance, same study
Penaeus japonicus	14	50	940	EC_{50}, 96h, active substance, same study
Penaeus japonicus	89	71	3,900	LC_{50}, 96h, active substance, same study
Stenelmiss sp.	85	99	148	EC_{50}, 96h, active substance, same study
Stenelmiss sp.	208	366	148	LC_{50}, 96h, active substance, same study
Trichocorixa sp.	21	63	56	EC_{50}, 48h, active substance, same study
Trichocorixa sp.	35	450	1,473	LC_{50}, 48h, active substance, same study

A.2.1. Possible future refinements in assessing the comparative toxicity of neonics

The exercise above to place the neonics on an equal "footing" considers only the relative toxicity of the different compounds. In the real world, however, the likely aquatic impacts will depend also on the ease with which residues enter the aquatic environment. The potential for pesticides to be found in surface runoff depends on their water solubility, ability to bind to soil, and persistence in soils. Pesticide industry scientists (Chen et al. 2002) developed a validated indicator of runoff potential called the Surface Water Mobility Index, or SWMI. This index ranges from 0 (for low mobility) to 1 (for high mobility). These index values are calculated in Table A.7 based on properties obtained from the Pesticide Properties Database. On that basis, at least three neonics, including the two main seed treatment chemicals (clothianidin and thiamethoxam), are expected to be more likely to run off to surface water than imidacloprid. Therefore, the higher toxicity of clothianidin would be further exacerbated and the lesser toxicity of thiamethoxam would not be as advantageous as suggested, given that it is the most mobile of the three.

Table A.7. Surface Water Mobility Indexes (SWMIs) for neonicotinoid insecticides based on an algorithm designed by Chen et al. (2002).[a]

Pesticide	SWMI Index
Acetamiprid	0.35
Clothianidin	0.66
Dinotefuran	0.85
Imidacloprid	0.56
Thiacloprid	0.30
Thiamethoxam	0.82

a Input data from Pesticide Properties Database at https://sitem.herts.ac.uk/aeru/ppdb/index.htm

A.3. Additivity or synergisms

Monitoring data make it clear that a compound-by-compound approach, as currently employed by American and Canadian regulatory bodies, is not tenable in light of the frequent detection of multiple residues across various aquatic ecosystems. Morrissey et al. (2015) similarly advocated for assessing summed residues, contending that toxicity benchmarks were proximate enough to warrant a joint toxicity benchmark.

Contradictory findings emerged from studies by Maloney et al. (2017, 2018a, 2018b) regarding compound additivity. While laboratory experiments on a chironomid species seemingly demonstrated a greater-than-additive effect with combinations of imidacloprid, clothianidin, and thiamethoxam, outdoor experiments in pond mesocosms yielded no evidence of synergistic effects among compounds. Nevertheless, impacts on chironomid emergence generally exceeded predictions from laboratory data, albeit with considerable variability among pond replicates, rendering interpretation challenging. Intriguingly, Bayer Corp., a major neonicotinoid manufacturer, had suggested potential synergistic action among several neonicotinoid insecticides, obtaining a patent on this discovery (Bayer Crop Science 2010).

In a seminal study published in *Science*, Schmidt et al. (2022) merged field observations from 85 coastal California streams with mesocosm testing of the dominant neonics, imidacloprid and clothianidin. The abundance of mayflies (all species combined) was evidently impacted by both compounds, with a 50% reduction observed at time-weighted average concentrations (over 30 days) of 1,050 ng/L and 1,350 ng/L for imidacloprid and clothianidin, respectively. Notably, examination of cumulative emergence over time suggested discernible effects at concentrations as low as 1 ng/L for clothianidin, thiamethoxam, and, to a lesser extent, imidacloprid (Figure 2 in Schmidt et al., 2022), representing levels significantly lower than EPA's current chronic benchmark for imidacloprid reviewed above.

Integrating their findings with existing chronic studies, Schmidt et al. (2022) derived chronic HC_5 values of 17 ng/L for imidacloprid and 10 ng/L for clothianidin but suggested that these values may not adequately preserve cumulative mayfly emergence, thus warranting a reassessment of neonic toxicity, as discussed earlier regarding time-weighted toxicity.

Through their experimental streams (mesocosms), the authors confirmed that imidacloprid and clothianidin exhibited greater-than-additive behavior, acting synergistically in many instances. Field samples revealed that total mayfly extirpation occurred at concentrations of imidacloprid or clothianidin that caused only a 50% decline in abundance with either compound alone in mesocosm settings.

Neonic mixtures were detected in 56% of streams, with at least one neonic detected in 72% of sampled streams (N=85). Summed neonic residues reached concentrations as high as 5,760 ng/L. Imidacloprid often dominated the mixture, yet dinotefuran was the most frequently detected, and thiamethoxam registered the highest concentration. The authors noted that at least one of the EPA benchmarks (see online levels in Table A.3) was exceeded in 28% of the samples. All samples were collected during April–June 2017 under low-flow conditions, potentially missing peak residue levels following rainfall, although they did cover the period when larval communities are well developed.

In the previous reviews and analyses referenced above (Mineau and Palmer 2013, Morrissey et al. 2015, Mineau 2019, 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. I stand by that assessment.

A.4. Structural issues persist in EPA's assessment of neonicotinoids in aquatic systems

In addition to the significant issues previously discussed, there remain fundamental problems with how EPA is assessing neonicotinoids in aquatic environments. These core issues have been highlighted repeatedly but have yet to be addressed or even acknowledged by EPA or other regulatory agencies.

The most critical issue is the ongoing failure to consider the time-dependent nature of neonic toxicity. Tennekes (2010) was the first to propose that neonics act as "one-hit" chemicals, exhibiting nearly perfect cumulative toxicity. This implies that a small dose can be as hazardous as a larger one if the exposure duration is extended. This concept has been reiterated multiple times, most recently by Sánchez-Bayo and Tennekes (2020). Neonic residues have been detected in watersheds for more than a year post-application. Consequently, even chronic toxicity benchmarks, which are based

on 21- to 28-day tests, are inadequate. Following this logic, impacts on aquatic life are expected at levels far below the established chronic toxicity thresholds. Furthermore, experimental evidence suggests that even brief pulses of neonics can result in delayed mortality in exposed aquatic invertebrates, an effect not captured by current testing protocols.

Both of these issues pose a significant challenge to the current assessment methods for neonics. However, to my knowledge, EPA and other regulatory bodies continue to disregard these findings. Despite having more than a decade to address these concerns, no action has been taken. The question remains: Why not?

Additionally, EPA continues to evaluate the toxicity of neonics to freshwater and saltwater organisms separately. Our 2013 report argued that the available science does not support this distinction. The perceived lower sensitivity of saltwater or brackish species is likely due to a lack of toxicity data. This oversight potentially places species-rich estuaries and other coastal areas at a much higher risk than currently acknowledged. In its recent assessments, the EU has placed much more stringent benchmarks on saltwater environments because of the paucity of data (e.g., SCHEER 2021 for imidacloprid). A safety factor of 10 was agreed on after the data for freshwater and saltwater organisms were combined. In contrast to the way North American regulatory bodies carry out aquatic protection, the EU applies the "precautionary principle" when data are lacking.

A.5. How is the aquatic risk of neonics currently viewed in the wider scientific community?

A notable analysis that closely followed our earlier report (Mineau and Palmer 2013) was the Worldwide Integrated Assessment of the Impact of Systemic Pesticides on Biodiversity and Ecosystems (WIA). This assessment, conducted by an international group of scientists, reviewed the extensive body of science on neonicotinoid insecticides available at the time. In their review of aquatic ecotoxicology (Pisa et al. 2015; Van der Sluijs et al. 2015; Pisa et al. 2017), they concluded that realistic levels of water contamination could lead to deleterious effects on the physiology and survival of a wide range of species in terrestrial, freshwater, and marine habitats. Chagnon et al. (2015) extended this analysis, suggesting that declines in emergent invertebrate prey due to insecticide use could plausibly cause population declines in insectivorous bird species.

Morrissey et al. (2015) conducted the first broad-scale quantitative risk analysis by comparing literature-based effect benchmarks with the growing body of information on residue levels in water bodies. They found that 81% of maximum and 74% of average individual neonicotinoid concentrations exceeded their benchmarks of 200 ng/L (acute) and 35 ng/L (chronic). They emphasized that the situation was likely worse because several neonicotinoids are often detected together, necessitating a comparison of summed concentrations with effect benchmarks. They concluded that both short-term and long-term impacts of neonicotinoids were occurring on a broad geographical scale.

Sánchez-Bayo et al. (2016) reached similar conclusions, stating: "*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*." (Sánchez-Bayo et al. 2016)

The most recent global analysis appears to be by Wang et al. (2022). They derived both acute and chronic benchmarks by generating species sensitivity distributions, combining toxicity data from all available aquatic taxa (algae, amphibians, crustaceans, fish, insects, molluscs, and worms). Their plotted values ranged over about six orders of magnitude. When chronic data were insufficient for a distribution, they used acute-chronic ratios to derive chronic toxicity data, a method we also employed in our earlier report (Mineau and Palmer 2013). While including all taxa increases data availability, it overlooks the different mechanisms of toxicity across groups, making it inappropriate to include them on the same plot. Nevertheless, their results are presented in Table A.8. Their ecosystem-wide HC_5 values under-protect sensitive groups like crustaceans and insects. Possibly for this reason, they recommend applying a safety factor of 5 to derive benchmarks from sensitivity distributions, a common practice among European regulators.

Table A.8. Ecosystem-wide derived HC5 values and proposed benchmarks by Wang et al. 2022.

Compound	Acute HC_5 (ng/L)	Chronic HC_5 (ng/L)	Proposed acute benchmark (ng/L)	Proposed chronic benchmark (ng/L)
Acetamiprid	3,310	NA	662	6.2
Clothianidin	8,940	39	1,790	7.7
Dinotefuran	23,400	NA	4,670	16.4
Imidacloprid	2,710	30	540	5.9
Thiacloprid	3,010	3	601	0.6
Thiamethoxam	23,000	78	4,590	15.6

Although the principle of a single, all-encompassing toxicity distribution as performed by Wang et al. (2022) has significant limitations, an interesting takeaway from this benchmark derivation is the similarity in the chronic benchmark among all but one compound, all within a factor of 3. The proposed value of 5.9 ng/L for imidacloprid is very much in line with existing European benchmarks (Table A.3, above), although the methodology is completely different. Thiacloprid stands out as much more toxic than the others. When comparing their proposed benchmarks with measured water concentrations reported globally, they found no acute risks (unsurprising since their method under-protects), but 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.

APPENDIX B: CURRENT EVIDENCE OF NEONIC CONTAMINATION MORE GENERALLY

B.1. Monitoring and study results continue to show broad contamination of the aquatic environment

EPA's 2016 review of imidacloprid (USEPA 2016) concluded that its levels frequently exceed thresholds at which aquatic invertebrate species are negatively impacted. The review indicated that several key taxonomic groups of aquatic invertebrates, not merely the most sensitive ones, are likely to be adversely affected by the concentrations currently measured in the environment. This concern is amplified by the frequent presence of other neonics in the same samples. EPA wrote: "*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 key components of a risk assessment. EPA scientists were encouraged by the fact that actual water measurements closely matched their modeled levels. They estimated that 60% of seed treatment applications, 90% of soil applications, and 100% of foliar applications of imidacloprid would result in surface water contamination levels exceeding the 10.0 ng/L benchmark.

Morrissey et al. (2015) summarized global data, demonstrating that aquatic contamination is inevitable given current usage patterns and the sheer volume of neonics in use. The following examples highlight some key studies published either before or after that review:

Contamination of wetlands is expected and can be "excused" when applications are directly into the wetland or onto seasonally drained areas. Evelsizer and Skopec (2018) reported high contamination in field crops in Iowa, while Hayasaka et al. (2019) found similar results in Japanese rice paddies. Samson-Robert et al. (2014) detected levels as high as 55,700 ng/L of clothianidin and 63,400 ng/L of thiamethoxam in puddles on seeded fields, posing clear risks to aquatic organisms in these seasonal wetlands and indicating significant exposure for both vertebrate and invertebrate wildlife.

The persistence and solubility characteristics of neonics, however, coupled with their extensive use in a variety of conditions, have resulted in widespread environmental contamination. Anderson et al. (2013) found levels as high as 225,000 ng/L of thiamethoxam in playa lakes in North Texas. Main et al. (2014) reported clothianidin values up to 3,100 ng/L and thiamethoxam values up to 1,490 ng/L in small wetlands near canola seed treatments. Schaafsma et al. (2015) measured up to 16,200 ng/L of clothianidin and 7,500 ng/L of thiamethoxam in ditches outside cornfields and 3,250 ng/L of clothianidin and 16,500 ng/L of thiamethoxam in puddles up to 100 meters from the fields. In a later study, Schaafsma et al. (2019) observed maximum concentrations of 6,950 ng/L of clothianidin and 2,630 ng/L of thiamethoxam in tile drain water, with median concentrations of 350 ng/L and 680 ng/L, respectively, in water receiving tile drain inputs. These findings were from fields with an estimated application rate of only 19 g/ha of active ingredient.

Miles et al. (2017, with a 2018 correction) detected clothianidin concentrations as high as 450–670 ng/L in small lentic woodland bodies of water in Indiana, far from monitored corn and soybean fields. These levels were higher than those reported in ditch samples nearer the fields. Cavallaro et al. (2019) reported values as high as 35 ng/L of clothianidin and 230 ng/L of thiamethoxam in wetlands within the canola-growing area of Saskatchewan, Canada.

Several studies have reported contamination levels far above benchmark levels early in the season, before any neonic use. For example, Schaafsma et al. (2015) found the highest levels pre-seeding, indicating year-round contamination. Extending exposure periods increases the risk of adverse effects, as toxicity is known to increase with longer exposure durations. Current assessments do not account for this, as chronic ecological impact studies typically last only a few weeks, whereas field data show that wildlife exposure periods span months to years. This prevents recovery of affected systems. Additionally, sublethal effects such as feeding disruption, behavioral changes, and delayed development have not been fully considered in the ecological assessments of neonics.

In previous reports, water monitoring data for New York State (Mineau 2019) and California (Mineau 2020) revealed frequent exceedances of aquatic toxicity benchmarks. Hoyle and Code (2016) arrived at similar conclusions. However, these analyses often miss the critical information of repeated exceedances at many sampling sites, crucial to understanding the full impact of neonics. This point was emphasized in Mineau and Palmer (2013) by reorganizing data from Starner and Goh (2012) in California watersheds. Mineau (2020) provided another example from California, showing that imidacloprid concentrations in Quail Creek between May and November seldom dipped below 500 ng/L, 50 times the 10 ng/L benchmark. This report shows the same pattern of persistence throughout the sample period – typically ice-free periods of spring to autumn. It is not surprising that there are increasing reports linking neonics to field impacts.

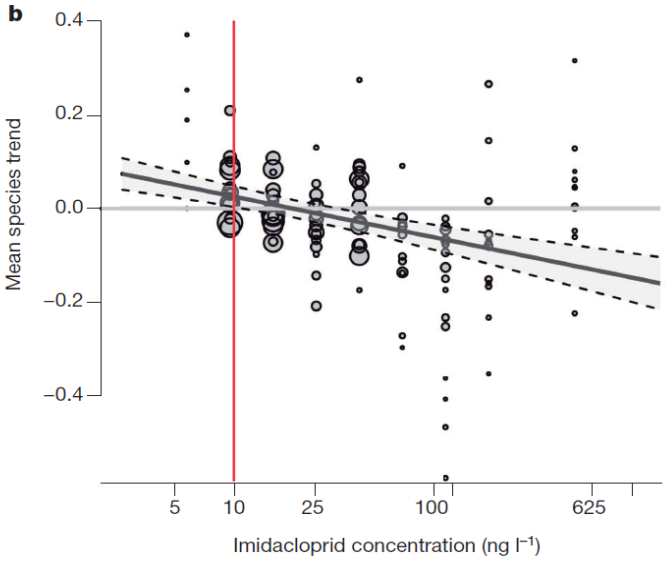
There is now incontrovertible evidence that pesticide loadings are a key factor in determining stream quality, as indicated by the presence of sensitive macroinvertebrates such as mayflies, caddisflies, and aquatic beetles (Reiber et al. 2020, Liess et al. 2021). Neonics, as the most important class of insecticides, significantly contribute to the degradation of freshwater systems worldwide and, likely, to estuarine and inshore marine environments. Associating specific compounds like neonics with biological outcomes such as insect emergence is challenging due to natural variability and difficulty in obtaining sufficient replicates in aquatic field studies. Despite these methodological challenges, evidence is accumulating that neonics are having clear negative impacts on aquatic ecosystems, paralleling documented effects in terrestrial systems.

B.2. Increasing evidence of reduced insect biomass and emergence as a result of neonic contamination

In Mineau and Palmer (2013), we reviewed an unpublished MSc thesis by Van Dijk (2010) from the Netherlands, which linked neonicotinoid contamination to reduced invertebrate numbers in Dutch canals. This work was later published as Van Dijk et al. (2013). Although Vijver and Van den Brink (2014) criticized the study for not accounting for other pesticide residues in the watersheds, Hallmann et al. (2014) indirectly supported Van Dijk's findings. They demonstrated that insectivorous birds declined in response to neonic concentrations (specifically imidacloprid) in water, and these declines did not occur before the introduction of neonics, despite the presence of other insecticides. Hallmann et al. (2014) predicted that regional bird declines would begin at water levels of imidacloprid of 200 ng/L or higher.

It is worthwhile to revisit Hallmann et al. (2014) in light of the continuing debate over benchmarks and safe levels in water (see Appendix A). The following figure (Figure B.1) is extracted from the article.

Figure B.1. Taken from Hallmann et al. 2014 with added line at 10 ng/L, the currently published EPA chronic benchmark.



Clearly, if ecosystem-wide impacts are to be avoided, a chronic benchmark close to 10 ng/L is indicated. It is rather encouraging (even though the evidence of environmental damage is disturbing) to see such concordance between laboratory-derived benchmarks and ecosystem-wide impacts.

Nowell et al. (2017) showed a relationship between mayfly abundance and maximum imidacloprid concentrations in streams in the Midwest. Yamamuro et al. (2019) documented the collapse of a smelt fishery in Japan due to neonic contamination from rice paddy culture, with spring plankton populations declining by 83% and the smelt harvest dropping from 240 to 22 tons. In June 2018, the total neonic concentration in a lake tributary was 72 ng/L, with imidacloprid, clothianidin, and thiamethoxam detected following rice planting.

Cavallaro et al. (2019) emphasized that agricultural landscapes already subject wetlands to various pressures, such as fertilizer and sediment runoff, that affect aquatic quality. They found that neonic inputs (primarily clothianidin and thiamethoxam, but also imidacloprid and acetamiprid) impacted insect emergence, habitat quality, and diversity. Their results showed that 73% of samples contained mixtures of neonics from different canola treatments.

Schepker et al. (2020) surveyed 26 wetlands in Nebraska during the spring of 2015, coinciding with the waterfowl spring migration rather than agricultural activities. They detected imidacloprid (max 5 ng/L) and/or clothianidin (max 16 ng/L) in 85% of wetlands, despite levels being below EPA benchmarks. They found that a buffer of more than 50 meters around wetlands reduced insecticide concentrations, but even at low levels, total neonic concentration negatively affected nektonic biomass.

Barmentlo et al. (2021) conducted an experiment with biweekly spikes of thiacloprid (100 to 10,000 ng/L) in ditches. Dragonflies, damselflies, and caddisflies showed reduced emergence following two 100 ng/L spikes, and total biomass and diversity were affected at 1,000 ng/L. Over 30 days, the two 1,000 ng/L spikes equated to a time-weighted concentration of 300 ng/L. The authors noted that changes in individual species often masked the broader disruptions caused by the insecticide, with some species benefiting from competition release as more sensitive species were impacted. They highlighted that these changes occurred at neonic levels commonly recorded worldwide and that they had likely underestimated the full impact due to the short study duration.

The work of Schmidt et al. (2022) in California showing current impacts on mayfly populations was reviewed earlier. In an ideal world, the good correspondence between laboratory-based predictions and the field should encourage regulators to impose more stringent regulatory benchmarks as well as restrictions and cancellations to reduce the environmental impact. This is clearly what has happened in Europe. In the United States and Canada, regulators appear to have paid no attention to the accumulating evidence. This evidence clearly underscores the significant impact of neonicotinoids on insect biomass and emergence. This, in turn, affects higher trophic levels, including insectivorous birds, thereby indicating a broader ecological disruption linked to neonic contamination.

B.3. Challenges in routine water monitoring of neonicotinoid levels

Routine water-monitoring exercises often fail to detect neonicotinoid levels as high as those reported in the scientific literature. This discrepancy arises because data from broad water-monitoring programs typically rely on “grab samples,” which can significantly underestimate peak surface concentrations of pesticides. Xing et al. (2013) demonstrated that relying on grab samples can lead to an underestimation by several orders of magnitude. This issue has been echoed by other researchers, such as Barmentlo et al. (2021), who emphasize the inadequacy of grab sampling in capturing peak pesticide concentrations.

The necessary frequency of sampling should be determined on the basis of the watershed’s size, as suggested by Crawford (2004). Crawford estimated that during runoff periods, samples should be taken at least 10 times monthly to ensure that peak measured residues are within a factor of 2 of the likely maxima. However, none of the Michigan sampling sites meet this criterion. This significant issue is often overlooked by regulators and state or watershed authorities, who typically focus on reporting the fraction of samples that exceed benchmark values without addressing the limitations of their sampling methodologies.

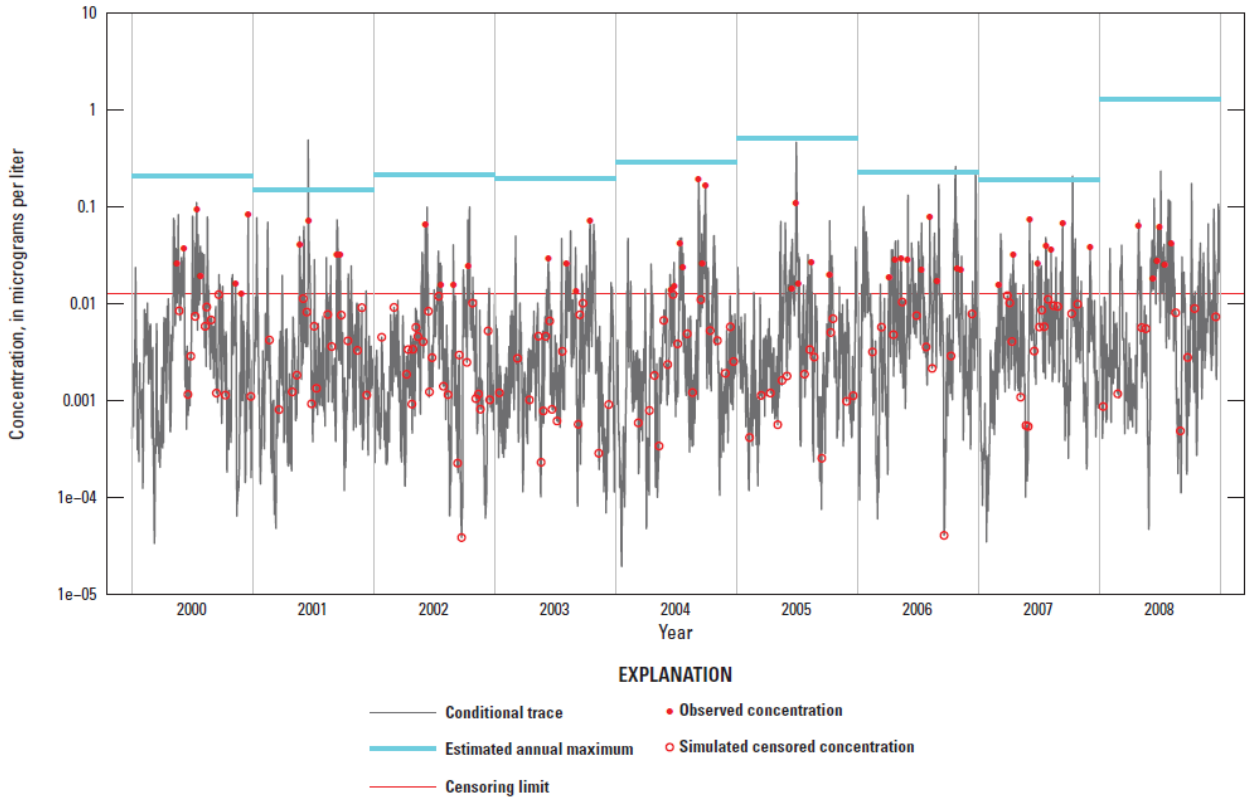
Accurate assessment of neonicotinoid contamination requires more frequent and methodologically sound sampling practices to capture true peak concentrations, thereby providing a more realistic picture of environmental exposure and risks.

B.3.1. The interpretation of measured water concentrations

USGS, recognizing the problems of occasional sampling as well as the problem of detection limits leading to heavily censored datasets, designed a sophisticated modeling approach (the SEAWAVE-QEX model – Vecchia 2018). The model may be difficult to use below 10 sampling visits per site,[17] which may rule out its application for Michigan sampling sites.

Figure B.2 is taken from Vecchia (2018) and shows the concentration of carbaryl in the Kisco River in New York State, with a detection limit just above 0.01 ug/L and peak residue detections around 0.1 ug/L. Estimated annual maximal values are commonly three to four times the highest observed value and sometimes more than 10 times the measured values.

Figure B.2. Plot taken from Vecchia (2018) to show the relationship between observed concentrations, simulated concentrations as a result of analyses below detection levels, and the estimated yearly maxima for carbaryl concentrations in the Kisco River, New York.



17 The full requirements are stated as follows: at least three individual years with six or more observations, 30% or more of which are uncensored, at least 30 observations for all years combined, and at least 10 uncensored observations for all years combined.

USEPA is currently considering how this model can be used in its drinking water assessments (USEPA 2020b and subsequent reviews by the Science Advisory Panel) but, to my knowledge, has not proposed applying any correction to water sampling data in order to assess ecological impacts.

Another limitation of most current datasets is that sampling is restricted to the summer months. Although this does cover most of the season of use of the pesticides, the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) Scientific Advisory Panel (SAP 2020), in its guidance to EPA, points out that "*in some situations, winter storms, especially the first flush after the dry period, often generate peaks in pesticide concentration*."

When it comes to using grab water samples to perform an ecological risk assessment, it is clear that using the raw data is fraught with problems and gives a false sense of security while under-protecting receiving environments. I believe we can demand better from EPA and other regulatory bodies.

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