



AN ASSESSMENT OF NEONICOTINOID INSECTICIDES WITH EMPHASIS ON NEW YORK: USE, CONTAMINATION, IMPACTS ON AQUATIC SYSTEMS, AND AGRONOMIC ASPECTS.

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FOREWORD:

This is an extended version of the report on this issue. A shorter version with highlights has also been published and distributed. See: *Mineau, P. 2019. Impacts of Neonics in New York Water. Their Use and Threats to the State's Aquatic Ecosystems. 18 pp.*

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1. GENERAL INFORMATION

1.1. INTRODUCTION

Neonicotinoids (neonics) are a relatively new class of insecticides now registered for a multitude of uses—from seed treatments to soil drenches to foliar-applied products. A major characteristic of the class is that they are systemic compounds; i.e., they have the ability to penetrate plant tissues to kill sucking or chewing insects. Unfortunately, the neonics most in use are also quite persistent in soils and, combined with their high water solubility, have now contaminated the broader environment. The fact that they are often used prophylactically, whether or not warranted by pest pressure, means that large quantities are used and they now dominate the insecticide market worldwide. They have been implicated in the collapse of honeybee populations as well as declines in a number of monitored insect species, such as pollinators.

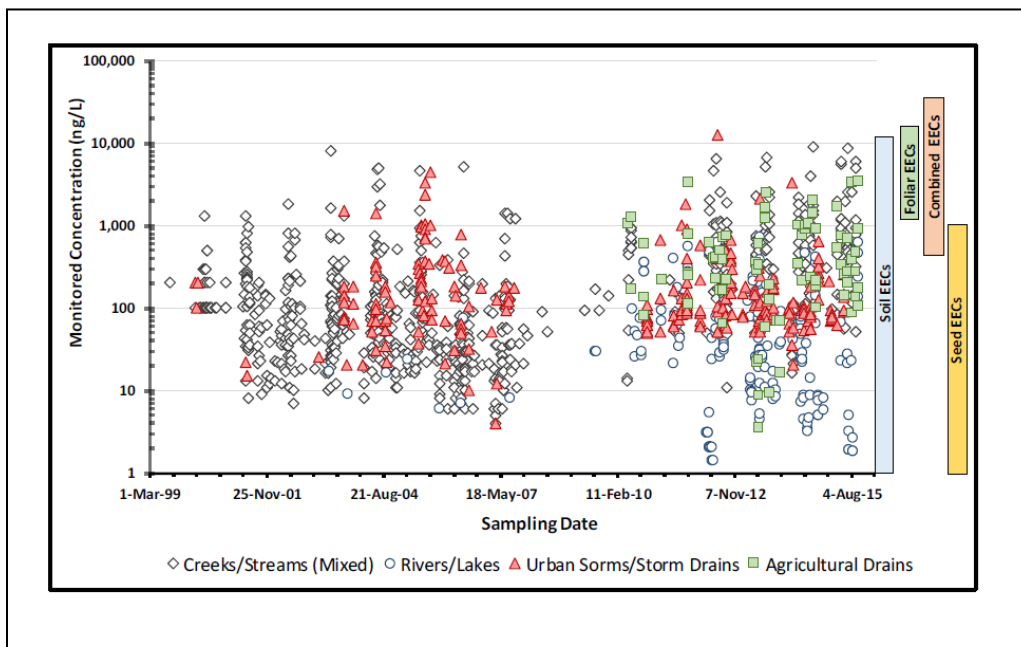
This report addresses their presence and impacts to the aquatic environment in New York State.

1.2. THE GROWING EVIDENCE OF NEONIC SURFACE AND GROUND WATER CONTAMINATION IN NORTH AMERICA

As the ability to analyze for neonic residues in water has grown, so has their frequency of detection. At this point in time, imidacloprid (the first registered neonic) is the one most often looked for, and found, in routine water monitoring programs. For the U.S. specifically, the U.S. Environmental Protection Agency (USEPA) has recently completed a review of available U.S. Geological Survey (USGS) and California surface water monitoring data (USEPA 2016). They reported that the detection frequency was high and that detected levels tracked reasonably well with predicted water concentrations modeled following various types of use patterns. Figure 1 taken from USEPA (2016) shows this very elegantly; concentrations of imidacloprid measured in routine surface water monitoring programs can be seen to reach 10 µg/L¹ or slightly higher. Based on modeling carried out by USEPA, the higher concentrations likely result from foliar spray applications.

¹ In order to reduce confusion, water concentrations as well as calculated benchmarks will be referred to as ppb or µg/L wherever possible.

Figure 1. Detected imidacloprid concentrations in U.S. surface waters from routine monitoring efforts relative to the range of modeled concentrations for different use patterns of the pesticide (USEPA 2016)



There is every reason to expect that, given their increasing use patterns, other neonics will similarly be detected with increasing frequency. A number of research studies and data reviews have shown that clothianidin and thiamethoxam, especially, are now being detected in surface water samples far and wide (Mineau and Palmer 2013; Main et al. 2014; Anderson et al. 2015; Morrissey et al. 2015; Miles et al. 2017, Struger et al. 2017; Bradford et al., 2018; Hladik et al. 2015, 2018). Similarly, neonics have been found to be ubiquitous in tap water in areas of use (Klarich et al. 2017; Sultana et al. 2018).

This is not surprising as several of the newer neonics are more likely than imidacloprid to run off to surface waters. 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) came up with 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 highly mobile). These indices are given in Table 1 based on properties obtained from the Pesticide Properties DataBase. Atrazine is provided as a comparison given that it is a ubiquitous contaminant of agricultural watersheds. At least three neonics (clothianidin, thiamethoxam, and dinotefuran) are expected to be more likely to run off to surface water than imidacloprid.

Table. 1 Surface Water Mobility Indices (SWMI) for neonicotinoid insecticides and atrazine based on an algorithm designed by Chen et al. (2002).^a

Pesticide	SWMI Index
Atrazine	0.76
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/>.

Ground water studies are fewer and smaller in scope, but evidence of contamination is clear (Mineau and Palmer 2013). Table 2 compares the potential of neonics to leach to groundwater as described by their Groundwater Ubiquity Score (GUS), a well-accepted predictive algorithm, developed by an industry scientist (Gustafson 1989), and based on soil persistence and carbon binding potential. Atrazine and aldicarb, historically important ground water contaminants of Long Island aquifers, are included in the table for comparison.

Table 2. Groundwater Ubiquity Scores (GUS) calculated by the Pesticide Properties DataBase^a and soil half-life for neonics as well as atrazine and aldicarb, two well-characterized groundwater contaminants

Pesticide	GUS Index ^b	Soil persistence (typical half-life in days)
Atrazine	3.2	75
Aldicarb	0.96	10
Acetamiprid	0.40	1.6
Clothianidin	4.9	545
Dinotefuran	4.9	82
Imidacloprid	3.7	191
Thiacloprid	0.14	15.5
Thiamethoxam	4.7	50

^a <https://sitem.herts.ac.uk/>

^b The GUS index is an indication of a compounds ability to leach and contaminate ground water. An index value above 2.8 denotes a compound with a high potential to leach.

It is clear that clothianidin and thiamethoxam, two commonly used neonic chemicals are more likely to be found in groundwater than imidacloprid, although most monitoring has been for imidacloprid only. Because clothianidin is the main breakdown product of thiamethoxam, use of the latter is expected to lead to long and persistent contamination of both surface and groundwater.

Where detected, neonics tend to be present for extended periods of time. For example, data by Starner and Goh (2012) from flowing water sources in California was re-analyzed by Mineau and Palmer (2013) to show the time aspects of sample collection. Water concentrations of imidacloprid were found to remain steady for periods exceeding three months (i.e., the entire summer) at all monitored sites. This persistence will have an important bearing on the choice of benchmark levels we choose to use (see below).

2. IS THERE A 'NO HARM' LEVEL OF NEONICS IN WATER?

2.1. ENVIRONMENTAL BENCHMARKS. HOW DOES THE USEPA COMPARE TO OTHER JURISDICTIONS?

To assess a chemical's possible impact on aquatic systems, we must estimate the water concentration of that chemical at which impacts to aquatic life are expected to occur. This is referred to as a "benchmark" value. Typically, a benchmark value is set by obtaining toxicity test data—hopefully, for a diverse range of organisms—and extrapolating from those data in order to derive a single metric that is sufficiently protective of the aquatic ecosystem as a whole. Because different jurisdictions approach this problem in different ways, different benchmark values are often derived for the same chemicals.

USEPA has long favored using a single test value for what they call the "most sensitive" species; i.e., the lowest acute or chronic toxicity value available to them at the time. This gives the appearance that all species will be adequately protected. However, this value often misleads because it depends heavily on the number of different taxa that have been tested, which is generally few. Even closely related species differ greatly in their sensitivity to pesticides—or chemicals more generally. The probability of truly finding the 'most sensitive' species, whether amongst aquatic invertebrates or other taxonomic groups, is much greater if many different species have been tested rather than a handful. Typical datasets, however, especially for newer pesticides, are generally too small (sometimes consisting of one or two species only) to offer any hope of finding the true "most sensitive" species. Therefore, even if contamination levels are kept under such a benchmark, damage to aquatic systems will often occur.

Recognizing this, most other jurisdictions have now adopted alternative strategies. One strategy places all available toxicity endpoints (e.g., all LC₅₀ values—the water concentration expected to kill half of the tested organisms) on a mathematical distribution and picks a single value based on the proportion of values expected to fall below this chosen value. The 5% tail of a distribution is often arbitrarily chosen as the

benchmark, although sometimes the 10% or 15% tail is used. Methods have been devised to approximate the results of a distribution analysis where there are too few values to plot a distribution. A parallel (and not mutually exclusive) strategy is to recognize that the 'most sensitive' species cannot logically be found, and that even a distribution analysis has limitations, especially when sample sizes are small; therefore, a safety or factor (either arbitrarily derived or based on similar datasets) is applied, either to the lowest value found in the sample of tested species or to a value derived by curve-fitting as described above. Use of a safety factor can also be a recognition that uncertainty derives not only from the difference in susceptibility between species, but also that organisms in the wild may be more sensitive than laboratory test organisms for a host of reasons. What is clear is that EPA's approach is the most problematic and least defensible scientifically.

2.1.1. Imidacloprid as a test case of benchmark determination at USEPA

The science behind a water quality benchmark can often initially be weak and change over time—raising caution about blind adherence to any calculated benchmark. USEPA's assessment of imidacloprid provides an excellent example of this, and shows how the pesticide's toxicity to aquatic systems has still not been satisfactorily addressed for marine environments. It took USEPA from 1994 to 2017 to derive acute and chronic freshwater (FW) imidacloprid benchmarks truly reflective of the available toxicological information and approaching those of other jurisdictions determined by more scientifically-defensible methods. Because USEPA's benchmark setting for "newer" neonics such as clothianidin, thiamethoxam, and others remain at a very early stage and employ the same inadequate process, they are likely flawed, as were the early imidacloprid assessments. Below, we will suggest a better strategy to assess the aquatic toxicity of these newer neonics.

The first USEPA environmental review for imidacloprid—then proposed for non-food crop uses only—took place in 1994 (USEPA 1994b). At the time, only two aquatic invertebrate species had been tested: a freshwater flea (*Daphnia magna* - EC₅₀ of 85,200 µg/L)^{2,3} and a saltwater (SW) mysid shrimp (*Americamysis bahia* - LC₅₀ of 37.3 µg/L). Between the two species there was more than 2,000-fold difference in toxicity—an indication that determining a benchmark imidacloprid level merited careful consideration. Using acute data for aquatic arthropods reviewed and vetted by EPA in 2016 (USEPA 2016), we can assess how this inter-species variance grew over time as more test species were

² EC refers to an 'effect concentration' rather than LC, the 'lethal concentration'. It has been long recommended that immobility or paralysis be used as a surrogate for lethality in toxicity tests. Such an effect as paralysis is clearly ecologically meaningful since the organisms cannot feed, swim or avoid predation; immobility is also easier to score than mortality. Toxicity data for aquatic invertebrates is interchangeably referred to as EC₅₀ or LC₅₀. Both are treated equally by regulatory bodies.

³ Subsequent tests on the same species have given LC₅₀ or EC₅₀ values ranging between 10,440 ug/L and 85,200 ug/L, showing the extent of test variability.

tested by different investigators. By 1997, the difference between most and least susceptible arthropod species was more than 27,000 fold among the 4 test species; by 2007 when EPA reviewed imidacloprid for a number of registrations, over 555,000 fold (11 species tested); and by 2017, date of the latest imidacloprid review, the spread stood at over 790,000 fold with a maximum of 36 species tested (28 named species). This progression demonstrates that finding the “most sensitive” species has more to do with chance than with good science. In the case of imidacloprid, the most sensitive of tested species to date goes back to 2007 (a mayfly species with LC₅₀ of 0.65 µg/L although the USEPA considers this value to be ‘qualitative’ and favours instead a value of 0.77 µg/L based on another mayfly species tested in 2013). USEPA may not yet have found the “most sensitive” species to imidacloprid as 36 test species still represents a very small portion of the invertebrate community in aquatic systems. Currently, a mayfly species is considered to be the most sensitive, but according to USEPA (2016) there are over 600 species of mayflies alone in the U.S., and only three species have been tested for imidacloprid.

In their initial review of imidacloprid, USEPA reviewers dropped their concerns about potential aquatic effects of imidacloprid when assured that the very limited proposed uses (commercial and residential landscapes, nurseries, and greenhouses) would reduce any exposure to aquatic species. However, that same year, USEPA received registration requests for imidacloprid applications on corn, potatoes, and apple. Despite the fact that USEPA currently assesses freshwater risk and saltwater risk using a separate set of toxicity endpoints, the USEPA reviewer (1994c) did consider the saltwater mysids as the ‘most sensitive’ aquatic species to be used in the risk assessment. In effect, given that concerns over acute exposure are raised when the ratio of exposure to toxicity is above 0.5,⁴ this was essentially setting USEPA’s aquatic invertebrate acute benchmark at 18.7 µg/L (LC₅₀ of 37.3 µg/L /2). A chronic benchmark at 0.16 µg/L was proposed based on the chronic NOAEL in the same saltwater species.⁵

In its 2007 risk assessment, USEPA formalized its acute and chronic aquatic toxicity benchmarks. However, it did not review the already extensive literature indicating the extreme toxicity of imidacloprid to certain groups of aquatic invertebrates, even though several studies in the open literature (2006 and older) showed acute toxicity levels as low as 1 µg/L. For the freshwater environment, USEPA scientists instead chose an acute value of 69 µg/L, the LC₅₀ value for a midge species, as the endpoint for comparison with predicted environmental concentrations.

Despite the fact that the water flea *Daphnia* had been shown to be a very insensitive species compared to other aquatic invertebrates, this was the only chronic data

⁴ This is referred to as the LOC (Level of Concern) by USEPA

⁵ The No Observed Adverse Effect Level, the effect here being length and weight of exposed organisms.

examined or required by EPA. However, when carrying out its assessments, USEPA abandoned the results of that test in favour of a value of 1 µg/L, obtained by applying a factor based on the acute-chronic toxicity differential seen in the saltwater mysid shrimp to the lowest acute FW endpoint. This was certainly far from ideal but to accept the Daphnia chronic test data would have meant setting a chronic benchmark higher than the acute one—a clear regulatory problem.

By 2017, USEPA could no longer ignore the large number of test results published in the scientific literature. Several extensive literature reviews and other regulatory agencies had developed aquatic toxicity benchmarks for imidacloprid that were radically different from those set by USEPA in 2007. In 2017, the USEPA found that the mayfly LC₅₀ value of 0.77 µg/L was the lowest quantitatively acceptable data point; placing the USEPA benchmark (given again a level of concern of 0.5 for aquatic systems at large) for peak water concentrations and acute effects at half of that concentration or 0.38 µg/L. The original industry mysid shrimp study (LC₅₀ of 33 µg/L) similarly anchors the saltwater acute benchmark at 16.6 µg/L, 44 times higher than the freshwater benchmark. On a scientific basis, the only thing that distinguishes saltwater from freshwater environments is not the sensitivity of its aquatic species, but merely the number and taxonomic distribution of species' test data (three crustacea and one diptera in saltwater). In fact, both the saltwater mosquito and blue crabs have lower LC₅₀ values than the mysid shrimp but those data are considered 'qualitative' by USEPA and are not used in setting the benchmark. This shows the arbitrary nature of USEPA's consideration of saltwater environments.

For chronic exposures (defined by USEPA 2016 as greater than four days, but typically resulting from studies of a few weeks duration), the minimum NOAEC values of 0.01 µg/L and 0.16 µg/L were retained for freshwater and saltwater respectively, the latter being the mysid shrimp—still the only chronic test data available to represent all marine arthropods. Because no concern for aquatic systems is raised with chronic values unless exposure equals the accepted NOAEC, these values are essentially comparable to other derived aquatic benchmarks.

In its latest assessment of imidacloprid, USEPA (2016) agreed with European and Canadian regulatory bodies that the use of higher tier studies (e.g., mesocosm or pond studies) proved problematic because sensitive species were insufficiently represented, producing an under-protective assessment. While favored by industry for imidacloprid (see Table 4), USEPA, as well as Canadian and European regulators have criticized this approach, as well as the industry's selective consideration of laboratory studies in the determination of an acute benchmark (USEPA 2016).

Tables 3 and 4 show a timeline of proposed acute and chronic benchmark levels for imidacloprid including the USEPA determinations reviewed above.

Table 3. A timeline of acute aquatic benchmark determinations for imidacloprid

Source	Benchmark (µg/L)	Justification
USEPA (1994b)	18.7	Based on the mysid shrimp – the lowest of FW and SW species multiplied by LOC of 0.5.
USEPA (2007) Freshwater	34.5	Lowest of three tests examined—to which a factor of two has been applied in keeping with the 0.5 LOC (Level of Concern) for a risk quotient.
USEPA (2007) Saltwater	18.7	As in 1994 above.
EFSA (2008) (Europe)	0.55	Lower of two species tested to which factor of 100 has been applied in keeping with Annex VI triggers for the Toxicity/Exposure Ratio.
RIVM (2008) (Netherlands – non regulatory)	0.2	Maximum acceptable concentration from short term exposure or exposure peaks and three-fold safety factor.
Nagai et al. 2012	0.43	HC5 ^a from a species sensitivity distribution (SSD) methodology which combines species within the same genus—also with 50% confidence.
USEPA (2012) (US – non regulatory)	35	Aquatic life benchmark online—accessed by Mineau and Palmer 2013—presumably same methodology as regulatory review.
Mineau and Palmer (2013)	1.01	HC5 (with 50% confidence) for acute exposure in crustacea.
Mineau and Palmer (2013)	1.02	HC5 (with 50% confidence) for acute exposure in insects.
Mineau and Palmer (2013)	0.22	HC5 (with 50% confidence) for acute exposure in all aquatic invertebrates (ignoring lack of normality).
EFSA (2014)	0.098	Median estimate of the HC5 of 0.49 based all insect studies (N=15) divided by safety factor of five. Incidentally, the lower 95% bound of the HC5 was also determined to be 0.098 µg/L.
Morrissey et al. (2015)	0.2	Lower confidence interval of HC5 from SSDs generated using 138 acute (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) Freshwater	0.36	Acute HC5 for 32 species tested.
PMRA (2016) Saltwater	1.37	Acute HC5 for six test species.
Bayer Crop Science (2016) (From EPA 2016)	1.73	HC5 after removal of several studies; rejected by USEPA 2017 because of biased acceptance of data points.
USEPA (2016) Freshwater	0.385	Based on quantitatively acceptable Mayfly study from open literature and factor of two.
USEPA (2016) Saltwater	16.5	Based on Mysid shrimp study. Unclear why slight difference with earlier derivations.

^a Hazardous Concentration⁵. An estimated concentration at which 5% of species will have achieved the endpoint – in this case LC₅₀.

Table 4. A timeline of chronic aquatic benchmark determinations for imidacloprid

Source	Benchmark (µg/L)	Justification
USEPA (1994b)	0.16	Lowest NOAEC of FW and SW species – Mysid shrimp.
USEPA (2007) FRESHWATER	1.0	Obtained with an acute/chronic ratio. (Using the usual chronic NOAEC for Daphnia would have meant accepting a value of 800 – much higher than the acute value).
USEPA (2007) SALTWATER	0.3	Based on the mysid shrimp. Unclear why different from the 1994 assessment.
CCME (2007) (Canada – non regulatory)	0.23	EC15 for the most sensitive of two freshwater species tested chronically to which a factor of ten has been applied.
EFSA (2008) (European Framework Directive)	0.2	NOAEC (0.6 µg/L) from a 21-day German microcosm study to which an assessment factor of three has been applied based on expert deliberations.
Dutch Regulatory Authority (2008) From RIVM 2008.	0.013	Maximum permissible concentration (MPC) for Dutch ecosystems.
RIVM (2008) (Netherlands)	0.067	Maximum permissible concentration for long term exposure derived from lowest NOAEC value and assessment factor of ten. This replaces the older value of 0.013 µg/L above.
USEPA (2012)	1.05	Aquatic life benchmark online - accessed by Mineau and Palmer 2013 – methodology uncertain.
Mineau and Palmer (2013)	0.029	Distribution analysis of NOAECs for chronic studies on seven single species and one species assemblage.
Mineau and Palmer (2013)	0.0086	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) (Netherlands)	0.0083	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 three applied.
Vijver and van den Brink (2014)	0.03	Proposed as relevant threshold based on chronic EC ₁₀ for two Mayfly species after the work of Roessink and colleagues.

European Food Safety Authority (2014)	0.009	Chronic HC5 of 0.027 based on ten studies from the literature. The assessment was based on the Netherlands analysis of the data. Experts agreed to apply a safety factor of three-fold.
Morrissey et al. (2015)	0.035	Lower confidence interval of HC5 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 (2015)	0.17	Following a review of five mesocosm studies. However, see comment about under-representation of sensitive species.
PMRA (2016) FRESHWATER	0.041	Chronic HC5 for ten species.
PMRA (2016) SALTWATER	0.33	NOAEC from only species tested.
Bayer Crop Science 2016 - As Moore et al. (2016)	1.01	HC5 from a selection of microcosm and mesocosm studies. Selection process criticised by PMRA and European Food Safety Authority.
USEPA (2017) FRESHWATER	0.01	NOAEC for mayfly study from open literature.
USEPA (2017) SALTWATER	0.163	NOAEC for industry Mysid shrimp study, the only endpoint considered. No change since 1994.

Several observations and conclusions emerge from our consideration of changing imidacloprid benchmarks over time. Most importantly, the USEPA acute benchmark for imidacloprid dropped 90-fold and the chronic benchmark 50-fold between its 2007 and 2017 assessments as USEPA began to consider data other than those supplied by industry. As USEPA concluded in its 2017 assessment, its currently chosen benchmarks—at least for freshwater environments—are very much in line with European, Canadian, and other credible independent benchmarks.

Unfortunately, USEPA's consideration of estuarine and marine waters always has been, and continues to be, woefully inadequate. USEPA admits to a higher "uncertainty" for saltwater benchmarks given the paucity of data, but a more credible scientific position would be to adopt the same benchmark values as for freshwater systems, given that the literature to date has shown this is scientifically justified. USEPA's guidance in this regard (i.e., to keep FW and SW species data separate) dates back to 2004. Shortly thereafter, Maltby et al. (2005) systematically explored the differences between toxicity estimates from distributions generated with data for freshwater and saltwater crustaceans for ten well-characterized insecticides. No significant differences were seen between estimates from these habitats. Even though saltwater species tended to be more sensitive, this was ascribed to the make-up of taxa most represented in the two habitats rather than any fundamental (toxicologically-driven) salt vs. freshwater difference. The continued USEPA approach to rely on a few arbitrarily-chosen brackish

or saltwater species to characterize the entire marine environment is scientifically unsound and needs to change in order to better protect waters in those States, such as New York that have a marine coastline and valued estuarine environments. At this point in time, the only scientifically-defensible strategy is to combine species and use the same benchmarks for all aquatic environments—a strategy in use in Europe, for example.

Similarly, as discussed in detail below, the currently-accepted benchmarks for the other neonics are in their infancy and suffer from the same issues: adherence to an inadequate methodological approach and failure to consider all available information.

2.1.2. Should we favour acute or chronic benchmarks?

In the case of neonics, especially, acute benchmark levels do not have much predictive value for several reasons. Regarding toxicity, neonics have shown extreme time dependence; i.e., toxicity increases dramatically as exposure duration increases. Widely reviewed (e.g., Morrissey et al. 2015), available research suggests that toxicity is essentially cumulative—meaning the length of exposure is inversely related to the effect concentration (e.g., Sanchez-Bayo 2009, Tennekes 2010). Relevantly, several studies have now shown that the high soil persistence of neonics combined with high runoff potential lead to prolonged watershed exposures ranging in weeks to months to possibly years. Findings of summer-long persistence by Starner and Goh (2012) in California flowing water sources were referenced above. Whiting et al. (2014, 2015) studied the consequences of a clothianidin seed treatment in corn. The rate of use was 0.25 mg/kernel or one-fifth of the allowable rate. Residues in runoff water were found right until the end of sampling—156 days after planting. Similarly, Schaafsma et al. (2015) measured levels up to 7.5 µg/L in ditches outside a seeded field and 16.5 µg/L in puddles situated in areas outside their field in central Canada; measurements were done pre-plant and therefore indicated contamination from the previous growing season. This argues for a chronic benchmark being much more realistic than an acute benchmark.

Pisa et al. (2017), concluding the second iteration of a worldwide multi-year assessment of neonics and other systemic insecticides, goes further:

“... acute LC50s or LD50s determined for short exposures (24 or 48 h) are irrelevant for risk assessments of these chemicals, because it is the long exposure to much lower levels of insecticide that really affects the survival of the organisms. It follows that protective levels for neonicotinoids cannot be achieved by setting a concentration benchmark, because the effects of neonicotinoids increase with exposure time and because of cascade effects within individuals.”

Concerns surrounding the adequacy of both acute and chronic benchmarks are not unique to neonics. Guy et al. (2011), in a comprehensive review of international

environmental water quality benchmarks, concluded that 88% of acute water quality benchmarks were insufficiently protective. This was based on a review of a large number of mesocosm and small pond studies and a comparison of standard toxicity test endpoints. In their analysis, chronic benchmarks fared slightly better, although even those underestimated impacts an estimated 63% of the time.

USEPA (2016), in their review of imidacloprid, point to other potential uncertainties, especially when estimating a chronic benchmark for that insecticide. In extrapolating from lab to field, most of these uncertainties would lead to underestimating the toxicity of the compound. In the chronic toxicity studies used to set the imidacloprid benchmark, initial measured concentrations gave rise to the benchmark whether or not degradation of the compound had occurred during testing. Following this principle is likely to lead to an underestimation of risk when (as is the case with imidacloprid) the breakdown compounds are less toxic to invertebrates than the parent molecule (USEPA 2016). The problem is particularly serious with several neonics because their primary breakdown happens through photolysis – exposure to light. For those compounds, rapid breakdown of the parent molecule is much more likely in the laboratory environment than in the field because tests are performed in clear water with strong illumination rather than in a turbid or shaded waterbody typical of the field.

2.1.3. What should be the proper benchmark levels to assess the aquatic toxicity of neonics other than imidacloprid?

As shown above, aquatic toxicity benchmarks for imidacloprid changed dramatically as more test data became incorporated in the calculation of those benchmarks. Table 5 shows the current (May 2019) benchmarks listed by USEPA for all of the registered neonicotinoid insecticides.

A quick review of the available toxicology data suggests that at least some of these benchmarks are not sufficiently protective. Thiamethoxam, for example, has an acute benchmark of 17.5 µg/L, already higher than the only LC₅₀ value available for mayflies (14 µg/L), an important component of aquatic ecosystems.

Table 5. USEPA aquatic freshwater benchmarks in effect as of May 2019

EPA 2019 Freshwater Benchmarks ^a	Acute (µg/L)	Chronic (µg/L)
Imidacloprid	0.385	0.01
thiamethoxam	17.5	0.74
clothianidin	11	0.05
thiacloprid	18.9	0.97
acetamiprid	10.5	2.1
dinotefuran	>484,150	>95,300

^a <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk> consulted January 2019.

Recent evaluations by Canadian regulatory authorities make use of the published toxicology information to derive freshwater benchmarks for thiamethoxam and clothianidin. Several potential benchmarks are proposed, but the most robust ones (based on a consideration of all the acceptable data fitted to a probability distribution) returned values very much at odds with the less-protective USEPA benchmarks for clothianidin and thiamethoxam.

Table 6. A comparison of current USEPA benchmarks with recent reviews of aquatic toxicity endpoints and proposed aquatic benchmarks by the Canadian Pest Management Regulatory Agency (PMRA) in µg/L

Compound	USEPA acute benchmark ^a	PMRA acute benchmark	USEPA chronic benchmark ^a	PMRA chronic benchmark
Imidacloprid	0.385	0.36 ^b	0.01	0.041 ^b
Thiamethoxam	17.5	9.0 ^c	0.74	0.026 ^c
Clothianidin	11	1.5 ^d	0.05	0.0015 ^d

^a <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-and-ecological-risk> consulted January 2019.

^b PMRA 2016.

^c PMRA 2018a.

^d PMRA 2018b.

The discrepancy between the USEPA and PMRA benchmarks for clothianidin and thiamethoxam is not surprising for the many reasons reviewed above; viz. imidacloprid's shifting assessment over time.

In our earlier assessment (Mineau and Palmer 2013), we proposed that the aquatic toxicity of thiamethoxam and clothianidin to aquatic insects and crustacea should be assumed to be similar to that of imidacloprid. We arrived at this conclusion after comparing toxicity tests performed on the same species with different neonics.

Similar conclusions were reached by Morrissey et al. (2015):

"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."

Indeed, the main difference between imidacloprid and other subsequently registered neonics is the richness of the data set; species by species comparisons show remarkably similar toxicity.

From this observation, Morrissey et al. (2015) concluded that scientifically-defensible benchmarks for clothianidin and thiamethoxam should more appropriately be 0.2 µg/L for short term exposures of all neonicotinoid insecticides combined or 0.035 µg/L for

chronic exposures in order to “avoid lasting effects on aquatic invertebrate communities.” These benchmark values were obtained by pooling all of the data for all neonicotinoids corrected by their molar mass. Of course, the benchmarks were heavily weighted to the imidacloprid data which made up more than half of the data points.

In this light, it is clear that the proposed USEPA benchmarks for clothianidin and thiamethoxam are untenable. PMRA’s analysis suggests that a scientifically-defensible chronic benchmark for thiamethoxam would be similar and, in fact, lower than that for imidacloprid. USEPA’s current value for a thiamethoxam chronic benchmark being 15-fold higher than its own clothianidin benchmark is also not defensible in a real world scenario given that clothianidin is the main breakdown product of thiamethoxam. In light of all this uncertainty, a defensible benchmark value for at least imidacloprid, thiamethoxam and clothianidin should be in the range of 0.01 – 0.04 µg/L. I propose to go forward in this report with 0.01 µg/L as the overall benchmark against which we need to compare measured water concentrations of all neonics. This value puts us in agreement with USEPA’s most recent benchmark for imidacloprid; in addition, it is appropriate to use the lower end of the range given that exposure duration in the environment (as reviewed earlier) far exceeds the duration of laboratory chronic tests.

While there are fewer data points for the other neonics, single species comparisons suggest that using the 0.01 µg/L benchmark for all neonics is a reasonable position. Indeed, the much lower value proposed by the Canadian authorities for clothianidin (0.0015 µg/L) is a concern in that it raises the possibility that impacts may occur at much lower concentrations than our chosen benchmark. We need not consider benchmarks calculated from acute toxicity endpoints because, as discussed above, they lack scientific validity and are not a meaningful metric for measuring neonics impacts to aquatic ecosystems.

Unfortunately, Canadian authorities, like USEPA, keep freshwater and saltwater species separate in their determination of a saltwater benchmark. This results in a clear under-protection of marine environments.

2.1.4. How do we account for mixtures of different neonics?

Strangely, in assessing the human and ecological risks neonics present, USEPA seems to ignore the fact that multiple neonic active ingredients are registered, which have similar modes of action, and which are frequently found together in water samples. In terms of potential human health impacts, USEPA has decided to treat all of the neonic active ingredients as unique compounds with unique modes of action:

“Unlike other pesticides for which EPA has followed a cumulative risk approach based on a common mechanism of toxicity, EPA has not made a common mechanism of toxicity finding as to imidacloprid and any other substances and imidacloprid does not

appear to produce a toxic metabolite produced by other substances. For the purposes of this tolerance action, therefore, EPA has not assumed that imidacloprid has a common mechanism of toxicity with other substances.” (U.S. EPA/OPP/HED 2010a, p. 40, referenced in USDA Forest Service 2016)

USEPA similarly stated regarding thiamethoxam that:

“... the Agency does not have data to indicate that thiamethoxam shares a common mechanism of toxicity with other chemical substances and therefore does not see a need for a cumulative risk assessment.” (USEPA 2011)

While many would not agree with this finding (e.g., USDA Forest Service 2016), the statement on imidacloprid might at least be a debatable one given slight differences in binding affinities of the different neonics to different cholinergic receptor subtypes. When applied to thiamethoxam, however, the statement is wholly unsupported given that another registered neonic, clothianidin, is thiamethoxam’s major breakdown product.

In an ecological context, there is no valid reason not to consider the registered neonics as a group. Morrissey et al. (2015) in their detailed review of all neonics, opined that they should be considered to have similar and, at least, additive toxicity to aquatic invertebrates. Accordingly, they proposed that their derived benchmark of 0.035 µg/L be applied to the sum of neonic residues. There is some indication from one of the manufacturers of neonics that this is a conservative strategy that may lead to under-protection. Indeed, Bayer Crop Science Inc. has argued that several neonics can act synergistically, and have filed a patent to that effect (Andersch et al. 2010). A synergistic effect is one where the effect is greater than predicted from simple additivity.

The need to consider the combined impact of all neonics together on aquatic systems is supported by recent research finding multiple neonics commonly in the same water samples in the field. Hladik and Kolpin (2015) in the first USGS national survey of neonics recorded that 26% of their samples contained more than one neonic; 11% of samples contained three or more neonics.

Finally, as reviewed by Morrissey et al. (2015) neonics have been shown to interact synergistically with other common pesticides or formulants. This also will underestimate the risk to aquatic ecosystems from a simple comparison of residue levels to benchmark levels.

2.2. DRINKING WATER STANDARDS

2.2.1. Are drinking water standards needed?

As early as 1994, even before imidacloprid was proposed for any food uses, USEPA reviewers were concerned about its potential to contaminate drinking water supplies:

"In summary, EFGWB⁶ is concerned about surface water and ground water contamination because [imidacloprid] has high water solubility and is persistent and moderately mobile based on K_d values. These are characteristics common to other pesticides that have been detected in groundwater. Repeated applications could cause saturation of soil sites thereby increasing desorption rates of future applications of this chemical increasing its potential for groundwater contamination. Also, if a heavy rainfall occurred following its application to a sandy soil with low organic matter content and the compound moved to an area below that of anaerobic microbial degradation, the resistance of [imidacloprid] to hydrolysis coupled with its mobility could cause ground water contamination. Our concerns are based on the results of laboratory studies; the field dissipation studies are ambiguous. . . . Because of the concern about the persistence and mobility of NTN and the possibility for groundwater contamination, EFGWB initially concluded that two long term field dissipation studies were needed. However, the need for these studies has since been re-evaluated with the determination that the studies would probably only provide information that would confirm that NTN is both persistent and mobile, which we already know. We have therefore determined that the long term field dissipation studies are no longer needed." (USEPA 1994b)

In their initial detailed review of the product and modeling of leaching potential, USEPA (USEPA 1994a) compared the potential for imidacloprid to leach into groundwater against other water-soluble insecticides with high leaching potential. In particular, imidacloprid was compared to aldicarb under a potato simulation using soil and rainfall data from Wisconsin. Imidacloprid was found to have a three-fold higher potential to leach to groundwater (6.1% of applied vs. 1.9% of applied) than aldicarb. These are very relevant results for New York State because of the long history of groundwater contamination by aldicarb on Long Island. Another simulation (tomatoes in California) indicated that 19.3% of applied imidacloprid would leach to groundwater.

Klarich et al. (2017) showed that neonics can survive standard water treatments. Sultana et al. (2018) found that, indeed, they are ubiquitous drinking water contaminants in agricultural areas. Clearly, the human population is potentially exposed to neonics in a chronic fashion – hence the need to derive drinking water standards.

⁶ EFGWB stands for the 'Environmental Fate & Ground Water Branch' of the Environmental Fate & Effects Division of the USEPA.

2.2.2. How are drinking water standards derived?

USEPA recently updated its guidance document for the derivation of benchmarks for human health assessment (USEPA 2017). The process starts by noting the reference doses (RfDs) and population adjusted doses (PADs) derived from a review of industry mammalian toxicology studies; lowest NOAELs are determined, to which certain safety factors are applied. These are chosen based on whether the benchmark applies to children, inter- and intra-species extrapolation, and completeness of the database and other sources of uncertainty. A modified process is used if a substance is deemed to be carcinogenic. Given discussions above concerning the length of exposure in surface water systems and the long persistence of residues once they enter groundwater, we will concentrate here on the chronic reference doses and the derived water concentrations relevant for chronic ingestion of residues.

The amount of daily water consumption is estimated to be 2.5 L/day for adults of 80kg. This is meant to be the 90th upper percentile of drinking water consumption for adults. (The calculation is also carried out for women of child-bearing age but this does not return a more protective benchmark.)

The calculation of reference doses also assumes that there will be exposure to the pesticides through other sources, notably diet. A Relative Source Contribution (RSC) of 20% refers to the proportion of the PAD remaining for drinking water after other sources have been considered.

Table 7 give the chronic or lifetime reference dose (rfD) or population-adjusted reference dose (PAD) as well as the water concentration that should not be exceeded in order to not exceed this lifetime dose.

The same industry studies were reviewed by other regulatory bodies, notably in California, Canada, and Europe. Concurrence amongst jurisdictions provides additional weight to the estimated safe levels in water even if the methods of deriving water concentration benchmarks vary.

Table 7. A summary of current Human Health Benchmarks for Pesticides (HHBPs) in drinking water^b

Active ingredient	Chronic or lifetime PAD (rfD) (mg/kg/day)	Chronic or Lifetime HHBPs (µg/L)	NOAEL in relevant chronic study (mg/kg/day)	Effect seen in study from which NOAEL established. (A complete review of the mammalian toxicology of the various pesticides is given in Mineau and Callaghan 2018.)
Imidacloprid	0.057	360	5.7	NOAEL for 2 yr. rat study. Thyroid effects at higher doses.
Clothianidin	0.098	630	9.8	NOAEL for 2 generation rat study. Effects seen include decreased body weight gain, delayed sexual maturation, decreased thymus weight in first generation offspring and increased stillbirths at higher doses.
Thiamethoxam	0.012	77	1.2	NOAEL for 2 generation rat study. Effects seen on sperm counts and testis weight in first generation offspring at higher doses.
Thiacloprid	0.004	0.8 ^a	1.2	NOAEL for 2 yr. rat study. Liver and thyroid histopathology, nervous system degeneration, and carcinogenicity seen at higher doses.
Acetamiprid	0.071	450	7.1	NOAEL for 2 yr. rat study. Liver and kidney toxicity, body weight and body weight gain effects seen at higher doses.

^a Different calculation to account for carcinogenicity of compound. The value of 0.8 µg/L is based on an estimated increased cancer risk of 1 in a million.

^b Obtained from <https://www.epa.gov/pesticides/updated-list-human-health-benchmarks-pesticides-drinking-water-available> accessed in January 2019.

The choice of NOAELs reported above met with general concurrence with the exception of acetamiprid. The EU (2004) identified a slightly lower NOAEL of 6.5 mg/kg/day based on a two generation rat study. However, both USEPA and PMRA identified the NOAEL as 18 mg/kg/day in the same study and the difference is slight and would not lead to a substantial difference in the resulting drinking water benchmark.

The California EPA (2018) calculated an imidacloprid acute reference water level for non-nursing infants of 283 µg/L, lower than the 360 µg/L level chronic calculated by USEPA. They recommended using this as the official California reference level for imidacloprid.

Clearly, the compound of most concern with respect to possible contamination of drinking water supply is thiacloprid because of the cancer risk it poses. The benchmark value for this neonic in drinking water, using USEPA's calculation for an increased cancer risk of 1 in a million results in a maximum permissible drinking water concentration that is a full order of magnitude below the next most toxic neonic: thiamethoxam.

However, drinking water standards do not take pesticide degradates into account and it is known that some of these are much more toxic to mammalian systems than the parent material. Recently, Klarich Wong et al. (2019) identified not just the known metabolites of imidacloprid (imidacloprid-urea and desnitro-imidacloprid) in drinking water supplies, but also four novel chlorinated products following water treatment. These compounds have not yet been characterized toxicologically and may present human health concerns.

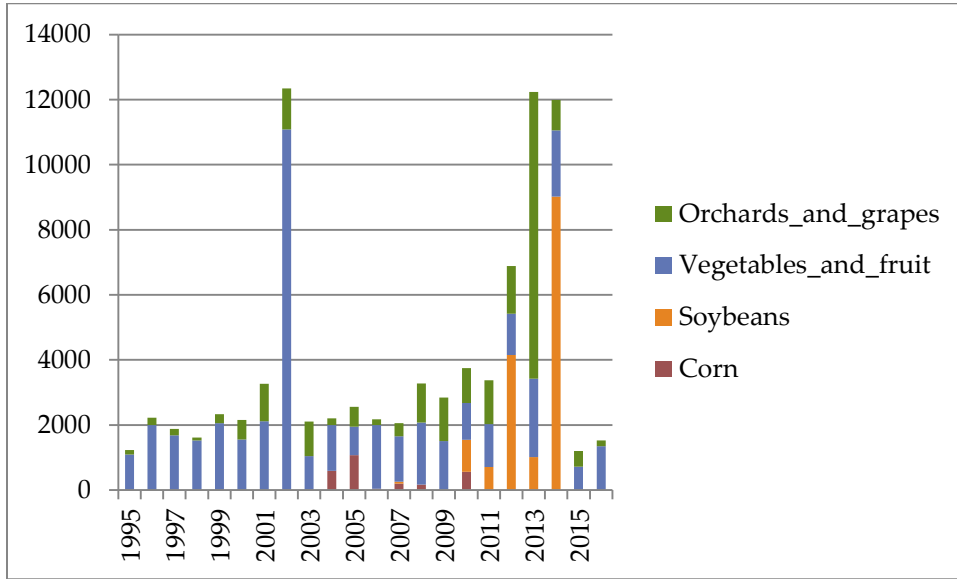
3. NEW YORK STATE AND NEONICS

3.1. HOW MUCH ARE NEONICS USED IN NEW YORK STATE?

USGS (Thelin and Stone 2013) developed a method for estimating pesticide use on a state by state and county by county basis by using confidential surveys of pesticide use patterns carried out on different crop types and then extrapolating these use rates by crop acreages grown in the different states and counties. Two estimates are provided: E-Pest Low and E-Pest High. The latter is especially useful in cases where use volumes are low in a given county and therefore use rate data are lacking. In those cases, E-Pest High interpolates from use rates in nearby counties rather than leaving a zero in the estimate, a much more realistic approach than assuming that no rate information equates to no use of the pesticide. I will use these interpolated results in this report.

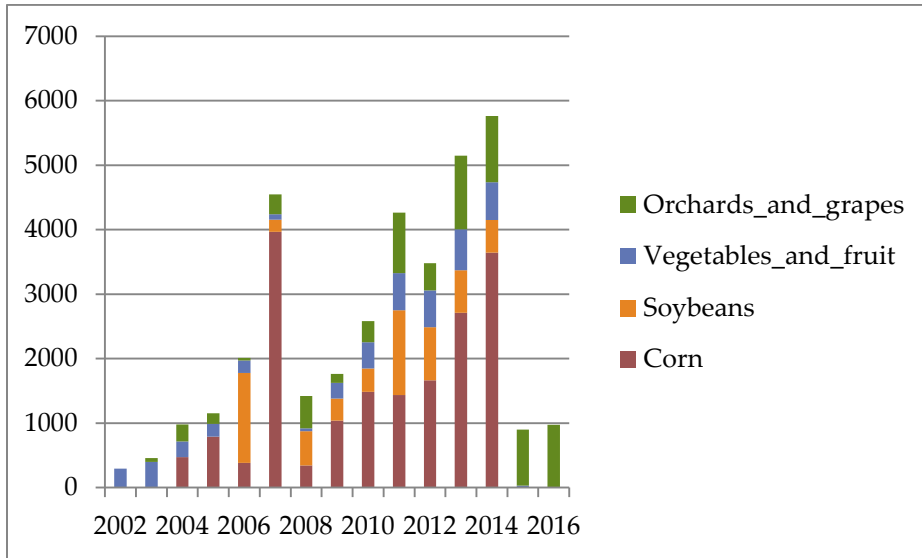
These estimated pesticide amounts reflect the agricultural use of the pesticides only—not domestic, landscape, or industrial uses—thereby significantly underestimating imidacloprid use given its extensive use for turf and ornamental applications. New York estimates from the USGS website are reproduced in Figures 2-7 below. The sudden drop in the estimated use of many active ingredients in 2015 is due to USGS no longer including seed treatments in their surveys as explained below.

Figure 2. New York estimates of imidacloprid use in kg^a



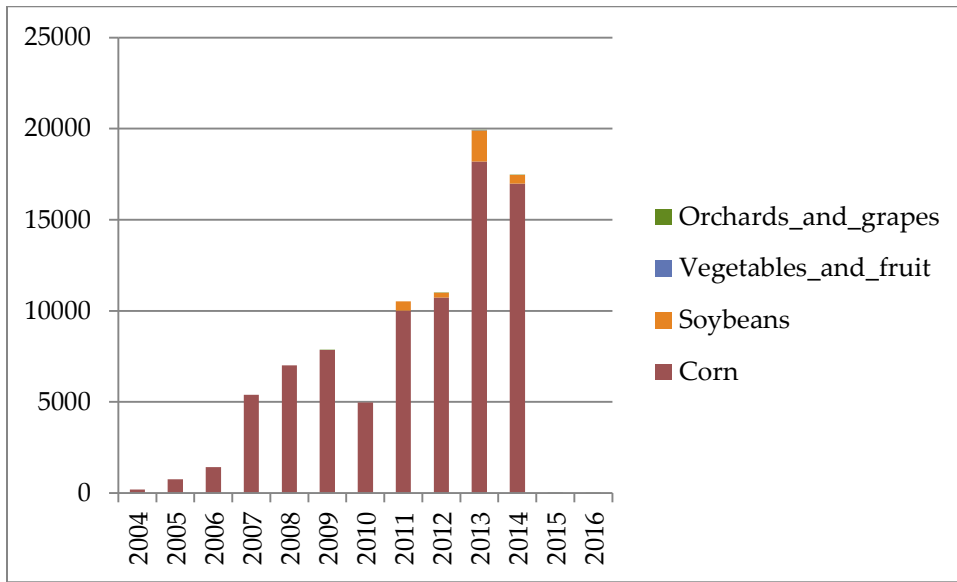
^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

Figure 3. New York estimates of thiamethoxam use in kg^a



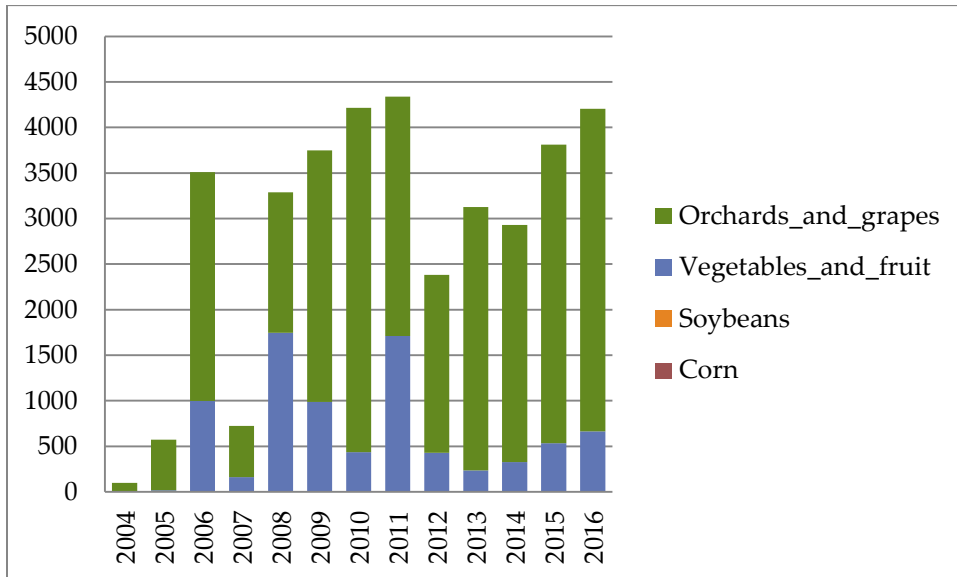
^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

Figure 4. New York estimates of clothianidin use in kg^a



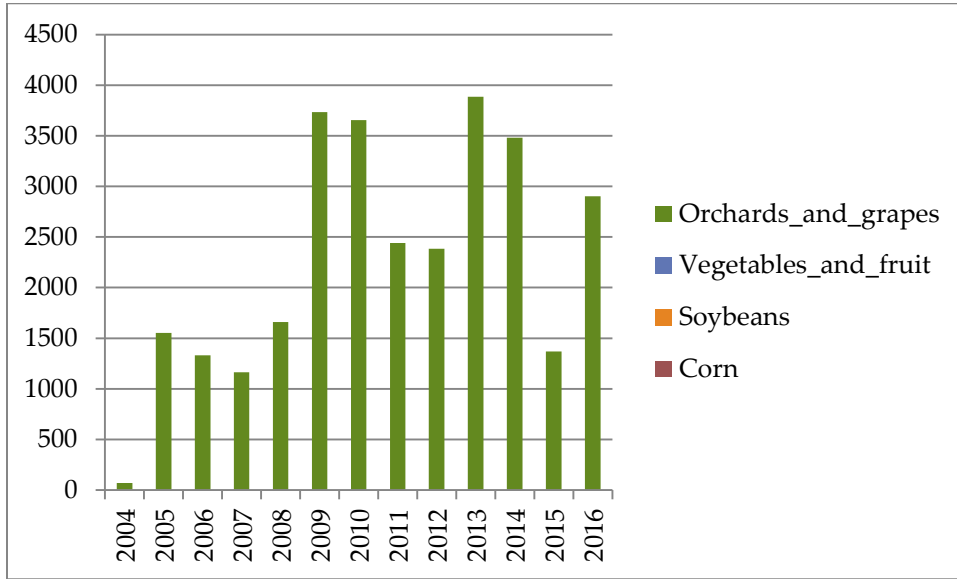
^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

Figure 5. New York estimates of acetamiprid use in kg^a



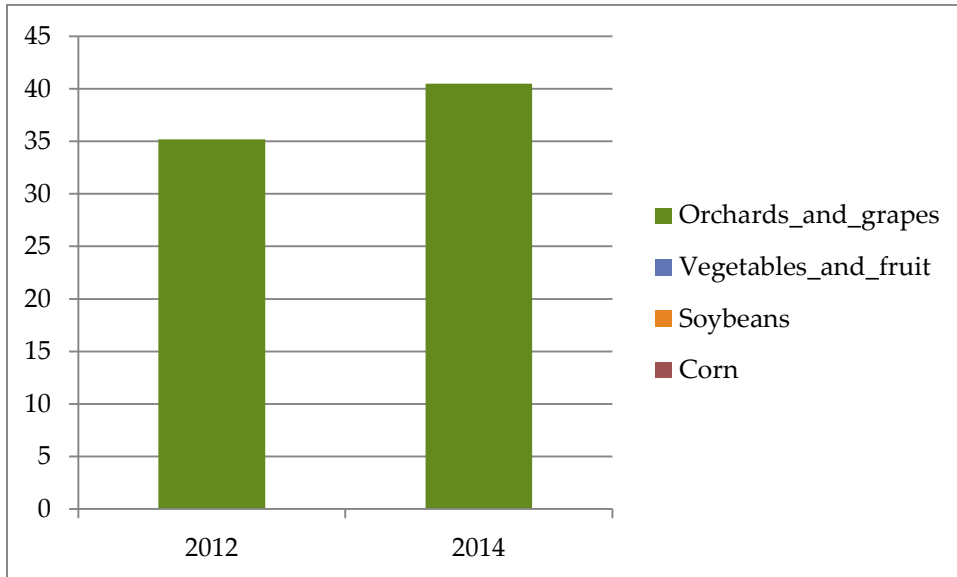
^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

Figure 6. New York estimates of thiacloprid use in kg^a



^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

Figure 7. New York estimates of dinotefuran in kg^a



^a Downloaded May 2020 from: <https://water.usgs.gov/nawqa/pnsp/usage/maps/county-level/>.

In 2015, USGS stopped including seed treatments in its estimates of pesticide use. This accounts for the sudden drop in estimated use seen for imidacloprid, clothianidin, and thiamethoxam. Although unfortunate, this change in procedure does allow us to estimate the relative importance of the seed treatments to overall use for the various neonic active ingredients. For this calculation, the nation-wide data were used in order to offer a better sample size. A reliable correction value is difficult to extract from the New York data alone, as the post 2015 levels drops to very low levels—even to 0 for clothianidin, for example. A correction factor based on national statistics is approximate as it is assumed that the same proportion will hold true for New York.

For example, a graphical extrapolation of the 2010-2014 nation-wide imidacloprid data suggests that, had seed treatments continued to be factored into the estimates, the total 2015 estimated use would have been approximately 2.4 million pounds.⁷ Without the seed treatments, the 2015 use was estimated to be 1 million pounds only. This suggests that, in 2015, approximately 58% of imidacloprid use would have been as a seed treatment.

Similarly, the steady rise of thiamethoxam use from 2004 to 2014 suggests that, in 2015, total use including seed treatments would have conservatively reached 1.5 million pounds. Without seed treatments, the 2015 use dropped just short of 0.4 million pounds. This suggests that seed treatments accounted for 75% of total thiamethoxam use in 2015.

For clothianidin, it is clear that ignoring seed treatment uses gives an even more biased estimate of its use. A graphical interpretation of the 2010-2014 time period suggests a total use of about 4.5 million pounds, but this drops to less than 0.4 million pounds when seed treatment uses are excluded—the highest differential of any neonic. Accordingly, we estimate that 92% of clothianidin total use comes in the form of seed treatments.

It is important to note that, relative to imidacloprid, clothianidin, and thiamethoxam, the use of the other three neonics is relatively small. In the case of acetamiprid and thiacloprid, seed treatments do not account for any measurable extent of use. Accordingly, future estimates for this active ingredient will not suffer from the same bias as the main three neonics unless use patterns change and new seed treatment uses are registered. Thiacloprid use nation-wide did drop measurably in 2015 but this neonic is mostly used in orchards and vineyards. The reduction in 2015 is therefore not attributable to seed treatment uses as was the case for the three main neonics described above. The use in New York did rebound in 2016 (Figure 6).

The New York data from the figures above was tabulated for two key years of estimated use: 2004, and 2014 (Table 8). As shown below, 2004 falls at the midpoint of

⁷ The national level maps are not shown here but can be obtained at: https://water.usgs.gov/nawqa/pnsp/usage/maps/compound_listing.php

intensive surface water sampling for imidacloprid in New York State and corresponds to the year with the highest reported frequency of occurrence of imidacloprid in state surface water samples. The year 2014 is the last year when USGS attempted to incorporate seed treatment pesticides in its estimates, as described above; it will be the reference year for calculating a total use estimate for New York.

Table 8. USGS estimates of agricultural neonic use for New York State in 2004 and 2014

Active ingredient	2004 quantity (kg)	2014 quantity (kg)
Acetamiprid	99	2,932
Clothianidin	196	17,491
Dinotefuran		40
Imidacloprid	2,202	11,995
Thiacloprid	70	3,482
Thiamethoxam	978	5,764
Total Neonics	3,546	41,704

The increasing use of neonics in New York State echoes the trends seen worldwide with respect to the increasing dominance of neonics in insecticide markets. USGS estimates suggest that neonic use in New York increased 12-fold in the 2004-2014 decade. These estimates also suggest that, for a decade at least, clothianidin has been the dominant agricultural neonic in New York, despite the fact that the state has effectively prohibited outdoor, non-seed treatment uses of that chemical (Serafini 2007). Because clothianidin has been largely used as a seed treatment, however, it is not captured by New York pesticide sales data – see below.

The New York State Department of Environmental Conservation (NYSDEC) also has pesticide reporting requirements maintained by Cornell University. Data collected prior to 2013 were reported by USEPA product registration number, making tabulation difficult without access to the machine-readable records. Starting in 2013, data were amalgamated by active ingredient. Several types of data are collected and the description of these data is given on the official website (<http://psur.cce.cornell.edu/Actives.aspx>).

Two of the data classes are deemed most useful for our purpose of building a full picture of neonic use in New York State (Robert Warfield, Cornell pers. comm.):

1. *Sales by commercial permit holders of restricted use pesticides or general use pesticides intended for agricultural crop production.* These data should best correspond with USGS estimates for agricultural use, with the exception of seed treatments, which are not considered pesticide applications by NYSDEC.⁸ Given the fact that a large proportion of seed treatments escape to the environment, either through runoff or in the course of seeding, the view that a seed treatment use is not a pesticide application appears to be based on convenience rather than sound science or logic.

2. *Use data by commercial applicators.* The data in these reports are from commercial applicators who are required to report each pesticide application, at least annually. Farmers are exempt from reporting.

The other two types of data collected by NYSDEC – *Sales data to end users*, and *Sales data to resellers* – are less useful in that they should be encompassed in the first two categories.

The 2014 data USGS estimates as well as the NYSDEC data are presented in Table 9.

Table 9. A comparison of USGS and NYSDEC data for the year 2014

Active ingredient	USGS estimate of agricultural use (kg)	NYSDEC - Sales by Commercial Permit Holders (kg)	NYSDEC - Pesticide Active Ingredients Applied by Commercial Applicators (kg)
Acetamiprid	2,932	541	1,774
Clothianidin	17,491	0	2
Dinotefuran	40	0	0
Imidacloprid	11,995	4,918	22,652
Thiacloprid	3,482	3,281	15
Thiamethoxam	5,764	1,957	828
TOTAL	41,704	10,697	25,271

⁸ Dan Wixted, Cornell University pers. comm.

The best estimate of agricultural use should be reflected by the "Sales by Commercial Permit Holders," which includes sales of restricted pesticides or pesticides intended for agricultural production. However, if farmers contract out their spraying to a commercial applicator, some of the agricultural use may appear there. The thiacloprid estimates (USGS vs. NYSDEC-recorded sales) are in good agreement without invoking applications by commercial applicators. This suggests that, for this active ingredient at least, there is not much involvement of commercial applicators on agricultural land. This is not the case for acetamiprid, however, meaning that either the USGS estimates are too high or there is a substantial contribution by commercial applicators on agricultural lands.

Above, we proposed a method for estimating the proportion of imidacloprid, thiamethoxam, and clothianidin applied in the form of seed treatments. This is based on graphical extrapolation of national USGS estimates from 2000-2014 (when seed treatments were included in the estimates) to 2015 (when seed treatments were no longer accounted for) and comparing this extrapolated value to the reported value for 2015. This provides a rough estimate of seed treatment use – 58%, 75%, and 92% for imidacloprid, thiamethoxam, and clothianidin, respectively.

Assuming that New York falls within the national average, these estimates allow us to correct the NYSDEC sales data to account for seed treatment use. For example, if thiamethoxam sales were corrected to reflect the fact that an estimated 75% of the use of thiamethoxam is not captured in the sales data, the corrected total would be approximately 7,800 kg or just 35% higher than the USGS estimated use. Likewise, the agricultural use of imidacloprid, once corrected for seed treatment use, could be estimated to be approximately 11,700 kg. This is in excellent agreement with USGS estimates. The good agreement between NYSDEC-recorded sales and USGS estimates for thiacloprid and imidacloprid, and the reasonably good agreement for thiamethoxam, suggest that the USGS estimates for clothianidin are also reasonable, despite the lack of NYSDEC-recorded sales. It is not surprising that no sales of clothianidin were recorded given that this chemical is almost exclusively used as a seed treatment.

One difficulty with sales data, of course, is that the use in any given year does not necessarily reflect use in the same year. However, over a number of years of survey, the supply and demand equations should equilibrate to reflect the turnover of various products.

We believe that the best overall estimate is a combination of the USGS and/or corrected NYSDEC estimates of sale for agricultural use. The non-agricultural use is best captured by the NYSDEC use data by commercial applicators. As mentioned, some of these applicators might be on contract to farmers, but, given the good agreement between USGS estimates and corrected NYSDEC sales data, and that imidacloprid

represents 90% of the neonics applied by commercial applicators, the survey likely captures the extensive turf, ornamental, and industrial site use of the pesticide rather than a major part of the agricultural use.⁹ The recent work by Nowell et al. (2017; but samples were taken in 2013) in the Midwest confirms that higher imidacloprid levels are now being seen in sites with urban inputs. Based on our assumptions and correcting for seed treatment applications, the only neonic for which application to agricultural land by commercial applicators appears to be happening to any significant extent is acetamiprid. Not knowing how much of the commercial applications overlap with USGS estimates for agricultural land leads to a higher uncertainty for this chemical.

Based on the logic detailed above, the best estimate of neonic use for New York State in 2014 is given in Table 10.

Table 10. Best estimate of outdoor neonic use in New York for 2014

Active ingredient	Best estimate (or range) of agricultural use (kg)	Best estimate of non-agricultural use (may include some application to agricultural land by commercial applicators for acetamiprid especially)	Best estimate of neonic use in New York State 2014 (kg)
Acetamiprid	541 - 2,932	1,774	2,315 - 4,706
Clothianidin	17,491	2	17,493
Dinotefuran	0 - 40	0	0 - 40
Imidacloprid	11,709 - 11,995	22,652	34,361 - 34,647
Thiacloprid	3,281 - 3,482	15	3,296 - 3,497
Thiamethoxam	5,764 - 7,828	828	6,592 - 8,656
Total Neonics	38,786 – 43,768	25,271	64,057 - 69,039

Despite the fact that clothianidin likely predominates in agriculture, imidacloprid remains the dominant active ingredient sold and used in the state. It should be noted,

⁹ Nevertheless, it is believed that applications by commercial applicators on farms is high (Dan Wixted, Cornell University, pers. comm.), although this is a general impression not specific to any particular product.

however, that the estimated 64-69 metric tons of neonic active ingredient used in New York State excludes homeowner and veterinary uses of products; this will lead to an underestimate of imidacloprid use given the plethora of domestic use products registered.

As of 2017, the main thiacloprid product (Calypso) appears to have been suspended. According to the NYSDEC list of registered products, it was never registered for use on Long Island.

3.2. WHAT IS THE EVIDENCE OF NEONIC WATER CONTAMINATION IN NEW YORK?

3.2.1. Surface water sampling

A download of all available water quality monitoring data was carried out in early February of 2019. The data were obtained through the National Water Information System of the USGS, accessed through <https://waterdata.usgs.gov/nwis> following guidance provided in Shoda et al. (2018).

USGS started reporting on imidacloprid residues in 2000, but with minimum reporting levels (i.e., quantification limits) of 0.1 µg/L only. In 2001, minimum reporting levels were dropped to 0.007 µg/L or just below our established threshold for biological effects (0.01 µg/L). Unfortunately, minimum reporting levels then changed over time so as to make detection more difficult. The minimum reporting level of 0.007 µg/L increased to 0.02 µg/L partway through 2004, and then to 0.06 µg/L partway through 2006. In 2006 and 2007, corresponding to this elevated reporting level up to 60-fold higher than the threshold for biological effects, results were often labeled as trace levels (t), below formal detection levels, and values were mostly given as estimates. Sampling effort dropped dramatically from 2008 to 2015 inclusive with few detections and an apparent unwillingness to estimate detected levels that were either trace or between detection and quantification levels. These years were combined into a single time period in the analysis below. From 2013 on, reporting levels were given as 0.011 µg/L on an interim fashion and then 0.016 µg/L once the use of a new calculation software was in place. This most recent reporting level is clearly inadequate in the context of harmful biological effects.

This variation of reporting levels makes it difficult to attach a great deal of importance to the proportion of samples showing detections. However, we can provide a rough estimate: In the years 2001 to 2007 and in 2016, the proportion of surface water samples analyzed from New York State that showed detections of imidacloprid varied between 15% and 50% of samples—without regard to minimum reporting levels. The proportion of positive samples (with imidacloprid detections) peaked at 50% in 2004.

The other reason the proportion of positive samples is not particularly meaningful is that we lack the context behind the samples: whether they originated from an area where pesticides were used or whether they were from agricultural, urban, or mixed environments. In addition, samples were taken at various times of the year and from various types of waterbodies, including large rivers where the dilution factor would be extreme.

In terms of understanding consequences to receiving environments, it is more meaningful to look at samples where imidacloprid was in fact detected, 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 are summarized in Table 11 below. They represent all positive samples reported for any given year, with several samples typically being reported for any given site (i.e., the values from which the tabulation was made are not strictly independent).

Table 11. A summary of New York State imidacloprid detections from USGS database

Year	Number of Detections	Mean concentration (µg/L)	Median concentration (µg/L)	High value recorded (µg/L)	Proportion of values above 0.01 µg/L	Proportion of values at least 10 fold higher than 0.01 µg/L
2001	50	0.197	0.132	1.30	98%	60%
2002	31	0.238	0.077	1.84	97%	45%
2003	93	0.271	0.074	7.94	99%	42%
2004	83	0.347	0.085	4.93	99%	60%
2005	63	0.373	0.092	4.64	100%	48%
2006	57	0.144	0.029	5.13	100%	12%
2007	49	0.146	0.033	1.40	90%	22%
2008-2015	8	0.025	0.018	0.073	62%	0%
2016	122	0.082	0.041	0.460	93%	26%

A number of observations can be made from the table. With the exception of the 2008-2015 period of reduced sampling intensity, it can be seen that, when detected, imidacloprid was present at levels above our impact benchmark level 90-100% of the time. A very high proportion of samples (up to 60% of samples in at least two years) had imidacloprid levels that were over 10 times the critical benchmark dose. This suggests that impacts to aquatic invertebrate fauna in New York State from imidacloprid alone have been substantial. Nowell et al. (2018) found that at the time of their study in 2013, imidacloprid accounted for the highest toxicity potential to benthic invertebrates in the Midwest, even though it occurred at lesser concentrations or frequency than many other pesticides. However, they did not analyze for other neonics.

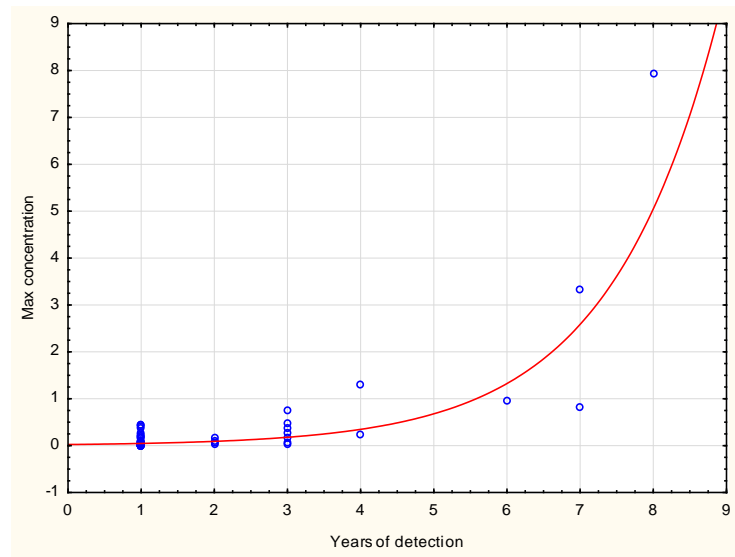
Routine water sampling may also underestimate the maximum level of contamination. Samples are often taken from large streams after much dilution has occurred, but impacts to aquatic life are expected where most of the aquatic productivity is taking place—in small drainage ditches and ponds bordering field areas to small feeder streams. Also, it is unreasonable to expect that grab or spot samples taken in the course of water monitoring schemes will necessarily coincide with peak concentrations of the various neonics in the monitored streams. Indeed, it has been shown that, even when taken weekly, water samples will underestimate peak concentrations by one-to-three orders of magnitude (Xing et al. 2015). Many of the sites chosen for sampling in New York State were sampled once only over the 15 year period we are examining. Even if sampling has been frequent enough to provide a “true picture” of expected residue concentrations (usually in the form of a distribution), there are difficulties in the interpretation of the results and clear biases when one tries to establish the proportion of samples that exceed benchmarks (Stehle et al. 2013).

Figure 8 supports the view that the ability to detect higher levels of contamination in surface waters is directly related to the intensity of sampling. Here, we have plotted the maximum observed concentration of imidacloprid (as high as 7.9 µg/L on one site) against the number of years imidacloprid was detected at the site.

These data suggest that high values of imidacloprid detected in routine water sampling are not anomalous, but simply reflect sites with more intensive sampling. If correct, this means that most sites with imidacloprid detections will, at some point, receive a high ‘slug’ of the insecticide capable of decimating its aquatic invertebrate fauna. We would be able to catch these moments in time given unlimited sampling effort, but this is clearly unlikely to happen.

Following a lull between 2008 and 2015, sampling for imidacloprid picked up again in 2016, although all samples sites were new locations.

Figure 8. Maximum imidacloprid concentration (in $\mu\text{g/L}$) recorded per monitoring site in New York State against the number of years imidacloprid was detected at the site



The number of samples analyzed for neonics other than imidacloprid in New York State are quite small. Based on the same search of USGS's National Water Information System over the same 2001-2016 period, a total of only 15 samples were analyzed for clothianidin and thiamethoxam; all samples originated between 2012 and 2016. Reporting limits were given as 0.0039 to 0.0062 $\mu\text{g/L}$ for clothianidin and 0.0034 to 0.0039 $\mu\text{g/L}$ for thiamethoxam. Unfortunately, only three of the 15 samples were taken during the May-June period corresponding to the seeding period. One of these samples, taken in early June from Fall Creek near Ithaca contained 0.020 $\mu\text{g/L}$ clothianidin and 0.0119 $\mu\text{g/L}$ thiamethoxam; the same sample also had trace levels of imidacloprid. An early May sample from the Genesee River in Rochester had detectable levels of both clothianidin and thiamethoxam just under reporting levels. Early May is still early for corn seeding, so those detections may have reflected prior year's use. A February sample also detected thiamethoxam at low levels.

The last three neonics, acetamiprid, dinotefuran, and thiacloprid were analyzed at three sites in New York State. Most of the sampling consisted of a monthly sample taken from the Genesee River in Rochester in 2015-2016. Minimum reporting levels were low (0.0032-0.0045 $\mu\text{g/L}$), but there were no detections from this limited sampling effort.

The low monitoring effort for neonics other than imidacloprid is unfortunate in light of the longer persistence and greater potential to contaminate surface waters from some of the other compounds. Hladik et al. (2014) found that both clothianidin and thiamethoxam occurred more frequently than imidacloprid in Midwest streams fed from agricultural areas.

The USGS water portal referenced earlier mentions integrated samplers and passive sampling disks being used for some of the neonics or their degradates, although it does not appear that any of these sampling strategies have been put in place in New York State.

A separate sampling effort was conducted in June 2015; the USFWS (Secord and Patnode 2018) took nine surface water and sediment samples downstream of crops typically treated with neonics—corn, soybeans, grapes, and apple orchards—and analyzed them for acetamiprid, clothianidin, imidacloprid, thiacloprid, and thiamethoxam. Aquatic invertebrate samples were also taken. Unfortunately, the detection limit was 3 µg/L or 30 times higher than the benchmark concentration determined above. No samples exceeded this detection limit. A second set of 17 sediment and 11 surface water samples was taken in July 2016 from streams downstream or in proximity to potato fields. Another five invertebrate samples described as ‘mostly crayfish’ were also taken. For those 2016 samples, they were able to drop the limit of detection (LOD) to 0.004 µg/L for the water samples, but the LOD remained at 3 µg/L for the other matrices.

Given the high LOD, only one of five invertebrate samples tested positive for neonics: 7.3 µg/L of thiacloprid.¹⁰ No sediment samples had detections above the LOD of 3 µg/L. Four of eight sampled locations tested positive for neonics. A summary of those detections is given in Table 12.

Table 12. A summary of neonic water detections in Secord and Patnode (2018)

Sample ID	Analytes	Concentration (µg/L)	Site description
NYP18	Imidacloprid	0.114	Salmon Creek 1, Wayne County, NY - Downstream of potatoes, corn, beans, orchards
	Acetamiprid	0.005	
NYP19	Imidacloprid	0.309 ^a	Salmon Creek 2, Wayne County, NY - Downstream of corn, beans, potatoes, orchards
	Thiamethoxam	0.005	
NYP20	Imidacloprid	0.006	Flint Creek, Yates County -

¹⁰ Imidacloprid was detected in one of 10 mussel samples from Pennsylvania in the same study at 6.3 ug/L.

			Downstream of potatoes, corn, beans
NYP23	Imidacloprid	0.018	Cohocton River, Steuben County, NY - Downstream of potatoes, corn, beans, wetland

^a Average of two measurements.

Three of the four sample locations with positive detects were above the 0.01 µg/L benchmark threshold. The combined concentration on one site exceeded the benchmark by a factor of 30. Half of the sites with positive detections had residues of multiple neonics.

3.2.2. Ground water monitoring

Based on the same search of USGS records, 1,023 analyses of groundwater looked for imidacloprid between 2001 and 2016. Analytical limits were as described above for surface water samples. Only 46 had detections ranging from trace levels at or just below reporting limits to a high of 5.3 µg/L. All but three of the samples had values above the 0.01 µg/L aquatic benchmark for ecological effects, but none approached the drinking water benchmark set by USEPA.

An exchange between NYSDEC (NYSDEC 2003, 2004) and Bayer Corporation (Bayer Corp. 2004) mentions the detection of imidacloprid in approximately 20 monitoring and private wells on Long Island. Imidacloprid was reported from well clusters down-gradient from farms and, in some cases, trees injected with imidacloprid, with the highest reported value at 0.0067 µg/L. There were concerns expressed over the fact that some wells with detections were well away from use sites, and others were deep (85-90 feet) community supply wells.

In 2014, NYSDEC summarized drinking water quality on Long Island based on analyses carried out by the Suffolk County Department of Health Services and Water Authority between the years 1996-2010 (Table 12). Imidacloprid was the only neonic analyzed. It was the sixth most frequently detected pesticide in Long Island groundwater, whether as part of the public water system or in private wells.

Imidacloprid urea, a degradate, was also detected in 6 monitoring well samples at a maximum of 1.3 µg/L. The guanidine and olefin degradates were not found.

Table 12. Imidacloprid detections in Long Island groundwater (in µg/L) from the Suffolk County Department of Health Services (1996-2010) from NYDEC (2014)

Drinking water supply				Monitoring wells			
Number of detects (wells)	Min.	Max.	Median	Number of detects	Min.	Max.	Median
549 (60)	0.1	12.9	0.4	341	0.04	407	0.7

A 2015 NYSDEC (NYSDEC 2015) report summarized ground and well water data from Long Island from 2001 to 2013. This report cited 0.2 µg/L as the lowest concentration detected (at odds with the previous report). Using that higher detection limit, the proportion of wells where imidacloprid was detected in the last year of sampling (2013) was 8.4% for monitoring wells and 7.5% for private wells with maximum levels of 6.2 µg/L and 2.8 µg/L, respectively. That same report pointed out that detection levels from 2010 to 2013 were lower than from 2005 to 2009.

The latest USGS information available for Long Island (2016-2017) provides results for 88 groundwater samples. Twenty-seven of those samples (31%) had detections of imidacloprid ranging from 0.005 µg/L to a maximum of 5.3 µg/L.

NYSDEC with the help of Cornell University investigated well water for pesticide contamination in several upstate New York counties from 2006 to 2010 (Whitbeck 2006, 2008, 2009a,b, 2010 – Table 13). Wells were chosen on the basis of intensity of nearby pesticide use and vulnerability. Imidacloprid was analyzed from year two of the program onwards but with a detection limit of 1 µg/L only. However, in the first year of the survey, samples were run with a commercial enzyme-linked immunosorbent assay (ELISA) for imidacloprid and related compounds (degradates as well as thiacloprid and acetamiprid) with a limit of detection of 0.07 µg/L and quantification limit of 0.2 µg/L.

Table 13. Summary of NYSDEC upstate New York sampling of wells

County	Collection times	No. samples taken	Detection limit (µg/L)	No. detects	Max. value (µg/L)	Reference
Schenectady	08-11/2005 and 03/2006	40	0.07	1	0.07-0.2	Whitbeck 2009a
Orange	06-08/2007	40	1	0		Whitbeck 2009b

Cayuga	05-09/2008	40	1	0		Whitbeck 2009c
Genesee	06/2009	40	1	0		Whitbeck 2010

3.2.3. Finished drinking water monitoring

Fourteen analyses of untreated or treated drinking water for imidacloprid were also reported from 2007 to 2011 in the USGS database. There were no detections.

Similarly, Suffolk County (2016) on Long Island reported on 1,500 analyses for imidacloprid in finished water from their various water distribution areas. There was only one detection, a value of 0.19 µg/L in one of 158 samples analysed from Distribution Area 30, the North Fork at the tip of Long Island, ranging roughly from Laurel in the west to East Marion on the east—an area supplied by 54 active wells. Unfortunately, detection limits were not provided. The report emphasized that, wherever possible, granulated activated carbon filtration and blending wells were being used to remove chemical impurities from water—explaining the low detection levels. No other neonics were tested.

3.3. NEONIC ENVIRONMENTAL BENCHMARK EXCEEDANCES IN CONTEXT

The various exceedances of imidacloprid levels for New York are not unusual. Based on studies carried out elsewhere, it is likely that exceedances of benchmarks for clothianidin and thiamethoxam would likely be recorded if those pesticides were adequately monitored.

The USEPA (2016) has already concluded that, nation-wide, imidacloprid levels are frequently above levels at which aquatic taxa will be negatively affected. Morrissey et al. (2015) following an exhaustive review of water measurements worldwide showed that their estimated reference levels (actually higher than the final 0.01 µg/L USEPA benchmark retained in this analysis) were often exceeded—and not just for imidacloprid. These results are also supported by other studies.

For clothianidin, for example, Main et al. (2014) reported values as high as 3.1 µg/L from water bodies in canola-growing areas following the use of seed treatments. Samson-Robert et al. (2014) found levels as high as 55.7 µg/L in puddles on seeded fields; Schaafsma et al. (2015) measured levels as high as 16.2 µg/L in ditches outside a seeded field and 3.25 µg/L in puddles as far as 100 m from the fields. Recent samples taken from a variety of waterbodies in crop and non-crop sites within an agricultural landscape in Indiana (Miles et al. 2017; with 2018 correction) detected concentrations of clothianidin as high as 0.45-0.67 µg/L in small lentic woodland bodies of water away

from the seeded corn and soybean fields. Levels in these wetlands were higher than those reported in any of the ditch samples taken nearer the seeded fields.

For thiamethoxam, Anderson (2013) found levels as high as 225 µg/L in the playa lakes of North Texas; Main et al. (2014) found values up to 1.49 µg/L from sloughs around canola fields; Samson-Robert et al. (2014) found levels as high as 63.4 µg/L in puddles on seeded fields; and Schaafsma et al. (2015) measured levels as high as 7.5 µg/L in ditches outside a seeded field and 16.5 µg/L in puddles outside their Ontario field. As reported earlier in this report, those two measurements were made pre-plant and therefore indicated contamination from the previous growing season. Higher levels were recorded in puddles within the field area.

As early as 2008 and 2009, Huseth and Grove reported levels of thiamethoxam in Wisconsin groundwater as high as 8.9 µg/L from its use on potatoes—a clear issue for any bodies of water recharged from groundwater.

Some of these exceedances have been reviewed and tabulated elsewhere; e.g., Morrissey et al. (2015). More recent reviews of exceedances can be found in PMRA's analyses: imidacloprid (PMRA 2016); thiamethoxam (2018a); and clothianidin (2018b).

Routine water monitoring exercises such as those carried out by the USGS (e.g., Hladik & Kolpin 2015) or state governments such as New York will not detect levels of neonics as high as those reported above. Despite the clear need for such geographically-distributed monitoring data, data collected as part of broad water monitoring exercises are severely limited in their ability to record true maximum levels, as discussed earlier. Additionally, extending the exposure period dramatically increases the risk of adverse effects because a clear relationship between toxicity and duration of exposure has been shown for several neonics; i.e., more toxicity expressed with increasing length of exposure. This has not been factored into current assessments; chronic ecological impact studies are carried out over the course of a few weeks only, while field data show that common exposure periods for wildlife are months to years, preventing any potential recovery of affected systems. In addition, sub-lethal effects such as feeding disruption, behavioral effects, and delayed development have also not been fully factored into the ecological effects of neonics.

3.4. EVIDENCE OF HARM FROM ENVIRONMENTAL BENCHMARK EXCEEDANCES

Not surprisingly, demonstrating a direct cause and effect between neonic use and the loss of invertebrate biomass at landscape scales is not an easy task. Aquatic systems are under threat from a host of stressors, and there are still too few measurements of neonics (even imidacloprid) to easily test cause and effect relationships. Where

imidacloprid or other neonics are measured, limits of detections are often above effect levels for sensitive groups of invertebrates (USEPA 2016).

Yet, USEPA (2016) has already concluded that imidacloprid levels are frequently above levels at which aquatic invertebrate species will be negatively affected. In fact, in its most recent aquatic risk assessment for imidacloprid, USEPA concludes that several key taxonomic groups of aquatic invertebrates, not merely the most sensitive ones, are likely to be adversely affected:

"... 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 is shown in Figure 9 reproduced from the USEPA assessment; toxicity benchmarks are compared to modeled water concentrations. USEPA showed in the same report that reported monitoring levels appeared to fit their modeled levels very well. They estimate that 60% of seed treatment applications, 90% of soil applications, and 100% of foliar applications of imidacloprid are expected to produce surface water contamination levels above the 0.01 µg/L benchmark—indicated in Figure 9 as the mayfly chronic value.

Nowell et al. (2017 – Figure 10) were able to show the consequences of imidacloprid detection on mayfly abundance in their monitored streams in the Midwest. As reviewed above, mayflies are known to be sensitive to neonics and are a key component of the benchmark levels established in all jurisdictions. The highest recorded concentration of imidacloprid in their study was 2.2 µg/L. Concentrations as high as 8 µg/L have been recorded in New York streams (above).

Van Dijk et al. (2013) concluded that neonics have affected aquatic invertebrate numbers in Dutch landscapes, although Vijver and van Den Brink (2014) criticized this conclusion for failure to consider residues of other potentially toxic pesticides, several of which were also present in the studied watersheds. Hallman et al. (2014) provides further evidence that we are seeing impacts from neonics much broader than on aquatic invertebrates alone. These authors analyzed insectivorous bird population trends in the Netherlands and showed a convincing correlation between neonic use and declining bird populations. Dividing the time period of their analysis into pre- and post-neonic exposure periods, they showed not only that neonic monitored concentrations explained bird declines, but that these site-specific declines were not seen before the introduction of neonics, despite the use of other insecticides of high aquatic toxicity. Neonic concentrations at which regional bird declines started being seen were estimated to be around 0.2 µg/L based on concurrent water sampling results.

Figure 9. Distribution of expected surface water concentrations of imidacloprid (plotted as exceedance frequencies) resulting from different use patterns showing where this distribution intersects various chronic toxicity endpoints determined for different freshwater taxa (From USEPA 2016)

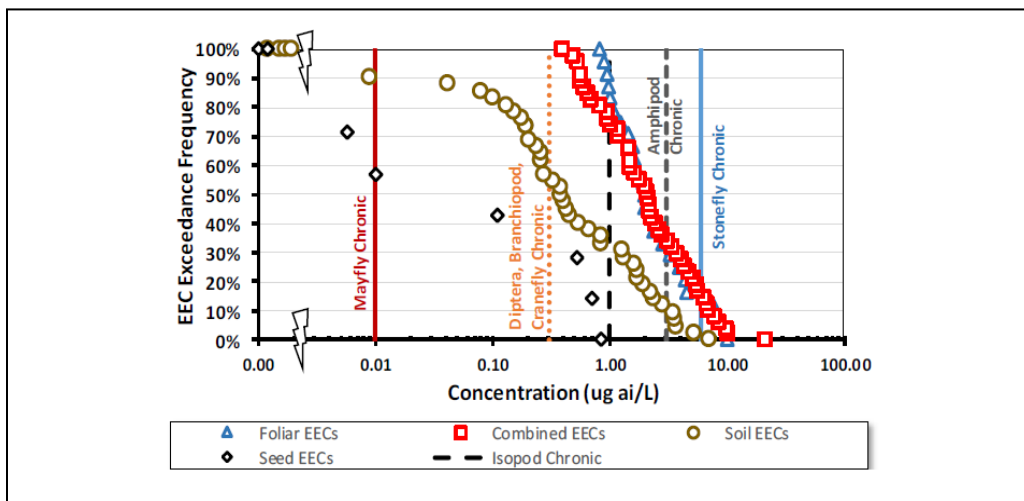
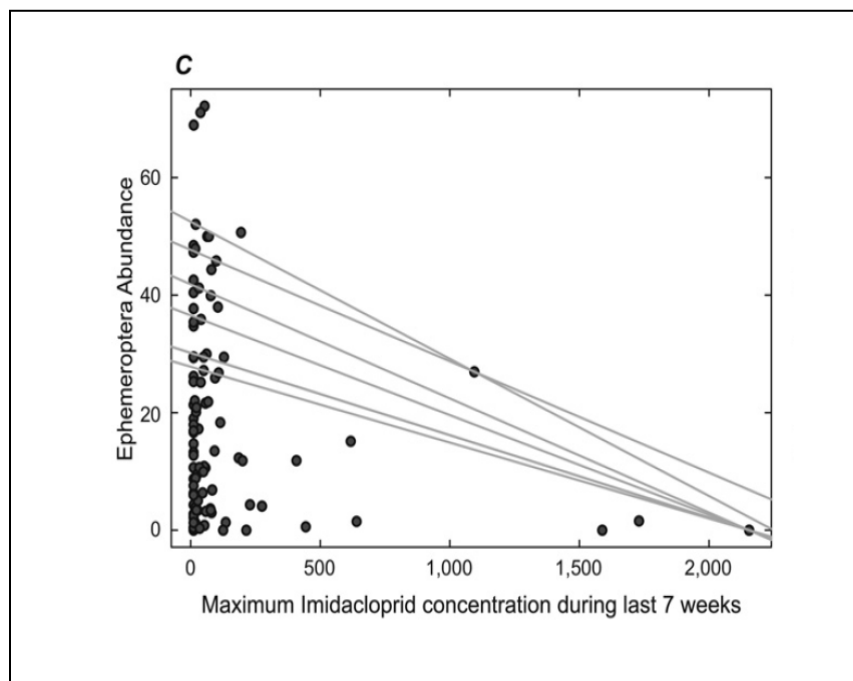


Figure 10. Relationship between mayfly abundance and maximum imidacloprid concentrations (in ng/L) in Midwest streams according to Nowell et al. (2017)



Beyond lethal effects, other reported effects on aquatic invertebrate larvae include feeding inhibition, reduced growth, mobility impairment, and delayed emergence (Goulson 2013; Morrissey et al. 2015).

There are more studies showing a relationship between neonics and terrestrial insect species. These are clearly relevant when considering the fate of emergent aquatic insects. Gilburn et al. (2015) using a similar analysis to Hallman et al. (2014) showed conclusively that neonics, and seed treatments in particular, are driving declines of butterfly species in the United Kingdom (U.K.). They showed that increasing population trends in several species were even reversed following the introduction of neonics; areas with low neonic use did not show the same extent of declines. Similarly, Forister et al. (2016) showed that, in California, neonic use was the best predictor to explain dramatic declines in butterflies beginning in 1997, after the pesticides were first introduced there in 1995. Other groups of insecticides were also examined, but did not show the same temporal association with butterfly declines. Incidentally, California, because of its good pesticide use registry, is probably the only state in the U.S. where this type of analysis could be performed. Forister et al. (2016) went on to show that the overall effect of neonics was equal to (but clearly additive to) that of habitat loss; butterfly species showing the strongest negative association with neonic use experienced the most severe declines. Woodcock et al. (2016) were able to show that the use of neonics in oilseed rape (canola) in the U.K. best explained extinction rates of wild bee species. Several other studies have now shown a correlative link between neonic use (specifically clothianidin, thiamethoxam, and thiacloprid) and the presence and viability of both managed and wild pollinator species as well as beneficial insect predatory and parasitoid species (see detailed review in Pisa et al. 2017).

In agreement with Vogel (2017), we can therefore conclude that, although insects have suffered from agricultural intensification and habitat loss generally, neonics in particular have had an important role in accelerating their declines. The evidence is especially compelling for pollinators because this is where most of the research has been concentrated. However, it is clear that many of the same findings of lethal or deleterious sub-lethal effects in pollinating species also apply to other insect species.

Scientists and experts of all stripes are now convinced of the broad impact of neonics on ecosystems. Curiously, USEPA (2008) in one of its early reviews of thiamethoxam predicted “structural and functional changes of both the aquatic and terrestrial ecosystems” from the use of this insecticide. This was an unprecedented assessment and prophetic warning. Others followed suit: Tennekes (2010b), a Dutch scientist and naturalist who predicted a “disaster in the making”; the “Task Force on Systemic Pesticides” comprised of a large group of independent scientists under the auspices of the International Union for the Conservation of Nature (IUCN) (van der Sluijs et al. 2015, Pisa et al. 2015, 2017); the European Academies Science Advisory Council (2015) made

up of 29 independent scientists nominated by their respective countries; as well as several other independent scientists (Morissey et al. 2015; Sánchez-Bayo et al. 2016).

Regulatory agencies including the USEPA, Canada's (PMRA), and European Food Safety Authority essentially agree that imidacloprid presents a high risk to the aquatic environment. EFSA found high acute or chronic risks for several crop scenarios including glasshouse use (EFSA 2014). PMRA recently recommended a complete phase out of all outdoor uses of clothianidin, thiamethoxam, and imidacloprid on food and feed crops including seed treatments and outdoor ornamentals due to the evidence of serious harm to aquatic species and ecosystems (PMRA 2016b; PMRA 2018a,b). USEPA's assessment of imidacloprid was summarised in Figure 9 above. USEPA's re-evaluation of clothianidin and thiamethoxam was not concluded at the time of publishing this report.

4. AGRONOMIC CONSIDERATIONS

4.1. THE RISE OF NEONICS IN AGRICULTURE

Douglas and Tooker (2015) offered the first detailed analysis of the changing landscape of pest control with neonics—between 2003 and 2011—especially when used as seed treatments. They projected a continued increase in use, both through application to more crop-hectares and more crops as well as an increasing per-seed application rate as a result of inevitable pest resistance issues. This per-seed increase in application over time was already evident from their analysis.

Douglas and Tooker (2015) found at least 500 registered agricultural uses for neonics although, by 2011, three field crops (maize, cotton, soybeans) accounted for approximately 80% of neonic use nation-wide. Imidacloprid was dominant until 2003 when clothianidin and thiamethoxam were added to the mix. Given the exponential increase of the use of these neonics between 2010 and 2015 (see Figures 2-4 for New York State), the current use totals are likely much higher.

Virtually all of U.S. corn (maize) is currently treated with neonic seed treatments; approximately a third of the soybean crop is treated—although it can be as high as 75% in some states; and 52-77% of the cotton crop (Douglas and Tooker 2015). These three crops alone represented about 42 million hectares of cropland in 2011. Most seed treatments are now applied by seed suppliers and they have begun seeing seed treatments less as a 'cost of production' and more as a 'profit center' in their own right (Douglas and Tooker 2015). With the right marketing, industry has been able to convince growers that the products are needed even when they are not. This has been facilitated by the USEPA which has waived any requirement for any "product performance data" (CFS 2014). Though the costs of these treatments are often passed

on to farmers, they may have been largely overlooked as higher demand for seed (e.g., from ethanol production) and the introduction of transgenic varieties were already driving seed prices dramatically higher. Neonic treatments, by comparison, were a relatively minor contributor to the total (Douglas and Tooker 2015).

Douglas and Tooker (2015) found that neonics accounted for 43% of the total mass of insecticide applied to maize; this figure dropped to approximately 22% for soybean, 27% for wheat, and less than 4% for cotton. These will be important figures to keep in mind while considering the likelihood that alternatives can be found in a scenario of forced use reduction. Douglas and Tooker (2015) believe, as many other specialists do, that there are opportunities to dramatically reduce the use of neonics through a more judicious framework of integrated pest management.

Unfortunately, the only means of tracking seed-treatment use—the USGS analysis reported above—stopped including seed treatments in the same year that the Douglas and Tooker publication appeared. Ignoring seed treatment uses gives a very biased underestimate of pesticide use. The U.S. Department of Agriculture has claimed a reduced dependence on insecticides in maize and that the U.S. is moving to integrated pest management when the exact opposite is true (Douglas and Tooker 2015). Indeed, the current trend in pesticide use is radically at odds with principles of integrated pest management espoused by Federal and State Agencies.

4.2. ARE SEED TREATMENTS PREFERABLE TO OTHER APPLICATION TYPES?

There is a general belief, promoted by chemical manufacturers, that incorporating a systemic insecticide into a seed treatment is preferable to a foliar insecticide application (e.g., Jeschke et al. 2011; Crop Life 2013). One reason cited is that seed treatments have fewer impacts on non-target invertebrates, including predators and parasitoids that are key allies in pest control. Industry has argued that, using a seed treatment, residues are confined to the growing crop and will not affect any insects that do not directly attack the crop.

However, this claim is incorrect. In a very large meta-analysis of randomized field studies carried out in North America and Europe, Douglas and Tooker (2016) found that natural enemies of crop pests were affected as much by neonic seed treatments as they were by broadcast applications of pyrethroid insecticides—pesticide applications not generally recognized as being friendly to natural predators.

USEPA (2016), based on their standard runoff models, estimated that runoff of imidacloprid from seed treatments was expected to be less than from foliar applications—both because of the lower application rate per hectare as well as the fact that seeds are buried below the soil surface. However, the analysis does not

consider the vastly greater area affected by a prophylactic seed treatment as compared to foliar applications, where a rational cost-benefit analysis generally calls for only a small portion of the planted area to be treated. In addition, USEPA (2016) warned that, in calculating runoff potential, they omitted an estimate of the effect of the dust generated during seeding. We know that this is a major contributor to the environmental contamination potential of seed treatment applications. Despite the existence of industry-based “stewardship programs,” there is no evidence that dust drift from seeding operations can be eliminated. To the contrary, the available evidence of widespread dust contamination from clothianidin and thiamethoxam seeding operations was obtained in the context of such programs (e.g., Schaafsma et al. 2015; Tsvetkov et al. 2017).

Tsvetkov et al. (2017), working near seed-treated corn fields in Canada, found that observed broad contamination of wildflower pollen lasted for the entire summer period. This result was all the more remarkable for the fact that seeding operations had made use of mandated “fluency agents” to try to reduce the problem of dust production. Hladik et al. (2016) also documented extensive contamination by thiamethoxam and clothianidin in native bee species taken from Colorado grasslands because of distant seed treatment uses.

Similarly, Long and Krupke (2016) found a high risk to insects using pollen from wildflowers in areas close to cropland where clothianidin and thiamethoxam were used as corn and soybean treatments. A host of other pesticides were found contaminating pollen sources, including fungicides expected to act synergistically with neonics. In this study, pollen was collected long after sowing in order to minimize the impact of dust contamination from the seeding. The authors suggested that the broad contamination of wild plant resources in the agroecosystem was likely from wind-erodible surface soils or from movement via runoff in surface waters followed by uptake in wild plant species.

These results suggest that runoff calculations from seed treatment applications have greatly underestimated their environmental contamination potential. As for the contamination of the terrestrial environment, it also isn’t clear that seed treatments result in less contamination. Mineau and Callaghan (2018) recently provided a systematic review of the expected contamination levels in non-target insects following the use of seed treatments and foliar applications of the main neonics. They found that, given equal application rates, the contamination levels from a seed treatment application would exceed those generated by a foliar application. However, it is clear that application rates from a seed treatment are generally lower than foliar application rates so this also must be taken into consideration. To arrive at a final estimate of contamination, Mineau and Callaghan (2018) calculated RUD (Residue per Unit Dose values) from an extensive search of the literature. The generally accepted RUD concept is that final residue levels will scale linearly to application rates. It isn’t a perfect

concept, as reviewed by Mineau and Callaghan (2018), but is still widely used in all regulatory evaluations. Thus, the RUD value is the expected residue level following a 1 kg/ha application of an active ingredient (1 lb/acre for U.S. regulators). To arrive at reasonable prediction of residue levels, the RUD value is then multiplied by the application rate as follows:

$$\text{Residue concentration (mg a.i./kg)} = \text{RUD (mg a.i./kg / kg a.i./ha application)} * \text{application rate (kg a.i./ha application)}$$

Based on their review, Mineau and Callaghan found that a 1 kg a.i./ha application of a seed treatment was expected to generate surface residue levels on insects of 409 ppm compared to 105 ppm for a foliar application. This is entirely because of the dust issue at seeding. Using Canadian labels for the main registered neonics, they were able to predict insect contamination levels following seed treatment or foliar (conventional spray or airblast) applications at some of the higher label rates (Table 14). American rates of application might differ slightly (given the hundreds of registered products and labels) but would be very similar.

Table 14. Estimated surface residues in terrestrial insects following foliar sprays and seed treatment uses of the main neonics based on Canadian registered uses after Mineau and Callaghan (2018)

Active ingredient	Application method	Maximum application rate (in mg a.i./ha) and crop details	RUD (mg a.i./kg for a 1 kg a.i./ha application)	Expected residue concentration on insects following an application (mg a.i./kg or ppm)
Clothianidin	Foliar spray	350	105	37
	Seed treatment	99 (corn)	409	40
Acetamiprid	Foliar spray	168	105	18
	Seed treatment	45 (canola)	409	18
Thiacloprid	Foliar spray	210 (fruit trees)	105	22
Imidacloprid	Foliar spray	330 (turf)	105	35
	Seed treatment	196 (corn)	409	80

Thiamethoxam	Foliar spray	700 (ornamentals)	105	74
	Seed treatment	42 (wheat)	409	17

The conclusion to be drawn from this exercise is that, generally, expected peak levels of contamination on terrestrial insects are nearly identical whether the neonic is applied as a seed treatment or as a foliar spray. An exception might be where foliar levels are very high as is the case for the use of thiamethoxam on ornamentals as shown in Table 14.

4.3. BENEFITS ANALYSIS FOR NEONIC USE

There have been several reviews of neonics' "cost-effectiveness," but one has to be careful in assessing a narrowly-determined claim of economic benefit. Neonics were introduced at a time when there were major pest resistance issues with predecessors: organophosphates, and, to lesser extent, synthetic pyrethroids. Any novel class of insecticide would have had some success against resistant pests.

Douglas and Tooker (2015) found that neonics supported a shift to 'insurance' pest management where transgenic crops and neonic seed treatments are deployed whether or not warranted by pest pressure. They cite a recent survey where 39% of maize growers did not have any particular pest in mind when applying treatment; for soybeans, this number rises to 47-65% of users.

Douglas and Tooker (2015) argued that neonic seed treatments were generally not efficacious against soybean aphids, the main pest now accounting for insecticide use in that crop. Even USEPA (2014) concluded that the benefits of neonic seed treatments in soybean were negligible in most cases. Moreover, Douglas et al. (2015) reported that soybean yields can actually be reduced by neonic seed treatments because of the disruption of natural pest control systems, in a manner similar to a pyrethroid foliar application (Douglas and Tooker 2016). Based on their 2015 analysis, neonic seed treatments have not provided consistent benefits in corn, either.

The Center for Food Safety (CFS) (2014, 2016) similarly provided an analysis of the lack of evidence for consistent benefits from neonic seed treatments. Their first report summarized 19 peer-reviewed studies. They found that eight studies reported no benefits and 11 studies reported inconsistent yield benefits. Benefits were questionable in the case of sunflowers, peanuts, cotton, and soybean especially. Indeed, like

Douglas and Tooker (2015), they singled out the use of neonic seed treatments in soybean as a clear aberration—where damage occurs at a crop stage where growth dilution renders the seed treatment ineffective and where there are clear management guidelines for scouting and treatment only when aphid numbers are above economic threshold. Other studies have shown disappointing results from neonic seed treatments; e.g., Difonso et al. 2015 for cutworm damage in dry beans.

Despite dire warnings from the European pesticide industry about the effects of a partial neonic ban there, European reviews and studies referenced by CFS (2014, 2016) carried out, in some cases, following the ban show that yield losses have not followed.

More recent research in this area has only strengthened the view that neonic seed treatments in soybean are not effective and that farmers who use them are essentially paying for a useless product. A 2017 joint effort by Purdue University, Iowa State University, Kansas State University, North Dakota State University, the University of Minnesota, South Dakota State University, and the University of Wisconsin (Krupke et al. 2017) confirmed the lack of any general benefit from neonic seed treatments in soybean.

The story is largely similar in corn. Alford and Krupke (2017) found that neonic levels remaining in plant tissue are too low to provide any lasting protection to the growing corn plants. Not surprisingly, North et al. (2017) in a large study of corn growers in the U.S. South found a net economic advantage of treatment in only one of four states studied (Louisiana).

As pointed out by Furlan et al. (2018), it is often possible to predict infestation levels, even for early-season pests, such as wireworms, and to restrict insecticide use to the portion of the crop that really needs it or to adopt other strategies, such as the “mutual fund” crop insurance approach used in Italy. These authors also review the numerous examples of pest resistance that have resulted from the prophylactic use of neonics.

An Australian study (Macfadyen et al. 2014) recently recorded minimal yield benefits from any insecticide treatment in wheat. They saw some evidence of increased pest damage in untreated fields, but this damage had no effect on yield. More generally, this shows that many efficacy claims based on pest damage surveys are insufficient to make the case for pesticide treatment. Benefits were also inconsistent or low in canola (oilseed rape). These authors concluded there is great potential for considerable reductions in current prophylactic treatments.

A British analysis of 11 years of oilseed rape production (Budge et al. 2015) found no overall effect of imidacloprid seed treatments on farming profits despite a clear indication that honeybee losses were related to regional seed treatment use.

Moreover, Hokkanen et al. (2017), working in Finland, found that the yield in insect-pollinated crops such as oilseed rape and caraway were inversely proportional to neonic seed treatment use. This raises an interesting possibility that ineffective large-scale treatments on major field crops such as corn, soybean, cereals, or oilseeds may cause production declines in other insect-pollinated crops being grown nearby. These potential collateral losses should be considered as a cost in cost-benefit analyses of prophylactic neonic use.

Clavet et al. (2014) provided data and reviewed the available evidence for the effectiveness of neonics or neonic/pyrethroid combinations to protect golf courses from weevil damage. They found that protection was so short-lived and inadequate that it did not warrant the risk of disruption in natural control agents.

All of the above research raises questions as to why neonics are so extensively used given the lack of clear agronomic benefits associated with the bulk of their use, and given the increasing evidence of substantial ecological damage in both aquatic and terrestrial environments. Based on USGS estimates, it is estimated that, nation-wide, approximately 70% by weight of all neonics used in agriculture are applied as corn and soy seed treatments—both questionable uses as reviewed above. For 2014 in New York State, the proportion of all neonics used in corn and soybean as seed treatments is estimated to be 73% of the total used in agriculture.

As shown above, every claim of effectiveness (e.g., Hummel et al. 2014 and rice water weevil control with clothianidin, thiamethoxam, and chloranthraniliprole seed treatments) needs to be analyzed carefully to ensure that the reduced pest counts actually lead to higher yields and that the cost-benefit equation (including the long-term agronomic effects through disruption of beneficial organisms) is accurately and objectively calculated.

4.4. THE NEED TO MOVE AWAY FROM PROPHYLACTIC INSECTICIDE USE.

As many authors have pointed out, even some narrowly-defined benefits (i.e., where the yield increase may justify the monetary cost of treatment) need to be assessed in a wider context of costs and benefits. This is not a new concept. As early as 1977, USEPA intended to follow this approach, although there is not much indication today that they have heeded their own advice (Pimentel and Burgess 2014). Many of the direct costs of using neonics have been identified—including the loss of honey bee hives or pollination services, the loss of natural predators, and others. Impacts to the natural environment have also been identified in the sections above.

The European Academies Science Advisory Council (EASAC 2015) reviewed the evidence for the wider agronomic benefits of neonics in the context of broader agricultural principles such as “key ecosystem services of pollination, natural pest control and soil ecosystems, as well as the biodiversity that contributes to such services.” While most of the research emphasis and discussion has been on the state of managed honey bees following the introduction of neonics, the European Academies scientists placed more emphasis on natural pollinators and potential biocontrol agents. They arrived at the following main conclusions:

“ 1. There is an increasing body of evidence that the widespread prophylactic use of neonicotinoids has severe negative effects on non-target organisms that provide ecosystem services including pollination and natural pest control.

2. There is clear scientific evidence for sub-lethal effects of very low levels of neonicotinoids over extended periods on non-target beneficial organisms.

3. Current practice of prophylactic usage of neonicotinoids is inconsistent with the basic principles of integrated pest management as expressed in the EU’s Sustainable Pesticides Directive.

4. Widespread use of neonicotinoids (as well as other pesticides) constrains the potential for restoring biodiversity in farmland under the EU’s Agri-environment Regulation.” (EASAC 2015)

Furlan et al. (2018) as part of the wider “Worldwide integrated assessment of the impact of systemic pesticides on biodiversity and ecosystems” arrive at a similar conclusion:

“ Regulators should realize that a more restrictive regulatory framework is required for more sustainable agricultural practices such as IPM, with a strong willingness to use (present or future) highly toxic pesticides only as the last resort.”

The call for a saner approach to pesticide regulation and, indeed, food production more generally is as clear now as it has always been. Notwithstanding some of the short-term gains in crop yields in response to technological advances in agriculture, the future of intensive agriculture as we know it is bleak; forward-looking scientists are calling for a more sustainable approach. The words of the Special Rapporteur on the Right to Food to the United Nations General Assembly (UN 2017) summarize this recommendation as follows:

“ Without or with minimal use of toxic chemicals, it is possible to produce healthier, nutrient-rich food, with higher yields in the longer term, without polluting and exhausting environmental resources. The solution requires a holistic approach to the right to adequate food that includes phasing out dangerous pesticides and enforcing an effective regulatory framework grounded on a human rights approach, coupled with a

transition towards sustainable agricultural practices that take into account the challenges of resource scarcity and climate change.” (UN 2017)

Other learned groups of scientists and commissions (e.g., IDDRI 2018, Lancet Commission 2019) have called for a complete re-structuring of our food production systems with a view to severely reduce or even eliminate a large proportion of current pesticide use.

Clearly, the bulk of current neonic uses—i.e., prophylactic uses that carry real costs to farmers, which have no or minimal tangible benefits, and which imperil the natural environment as well as our long term ability to farm—should be among the first to be eliminated.

5. CONCLUSIONS

We can draw a number of conclusions from our analysis of neonic use and presence in New York State:

- About 70-76 U.S. tons (64-69 metric tons) of neonic active ingredient are estimated to be used annually. Neonic use increased exponentially over the last decade.
- Imidacloprid has the largest volume of use, but the other four USEPA-registered neonic active ingredients (thiacloprid having been suspended in 2017) combined make up a similar quantity. Clothianidin is the likely dominant neonic in New York agriculture.
- Imidacloprid residues are often found in surface water samples. Where they are found, they are found above the biological threshold of 0.01 µg/L more than 90% of the time, and frequently at levels 10 times the threshold.
- High recorded values (e.g., 6-8 µg/L) tend to be from sites where imidacloprid has been detected over multiple years. This suggests that more frequent/intensive monitoring would uncover other instances where imidacloprid can reach such highly damaging levels.
- While the aggregate neonic presence in water is the relevant measure of concern, imidacloprid data alone indicate a very high probability that neonics are causing ecosystem-wide damage—including depletion of aquatic invertebrate populations as well as possible harms to consumer species, such as birds, fish, and mammals.

- Some of the other neonics, notably clothianidin and thiamethoxam are more prone to runoff than imidacloprid. More monitoring for these products in regions of use is warranted, although research to date already suggests that they will be present in surface waters at biologically damaging levels.
- The conclusions of independent researchers and of the regulatory community appear to be unanimous: Use of neonics entails an inevitable loss of invertebrate life in both terrestrial and aquatic systems. These effects, in turn, lead to ecosystem-wide perturbations affecting consumer species such as insectivorous birds, bats, fish, and other vertebrates.
- As for drinking water, New York should monitor water sources for the presence of all neonics (down to detection levels of 0.01 µg/L or lower), especially where surface waters are used for drinking water and on Long Island or other areas with heavy neonic use and vulnerable aquifers.
- Benchmarks set for drinking water safety are currently set very high for all neonics, except thiacloprid, and are not apparently being exceeded. It is unclear whether recent research on the toxicity of degradates and chlorinated neonic by-products will change this going forward.
- Research has shown conclusively that, even when benefits are narrowly defined, neonics are rarely cost-effective for their main uses; e.g., corn and soybean seed treatments. Yet, seed treatments in corn and soybean make up about 73% of agricultural neonic use by weight in New York State. There is reason to think that because of their clear impacts on natural enemies, neonic use is actually injurious to some cropping systems.
- Despite mounting evidence to the contrary, neonic manufacturers continue to mislead the public (including the farming community) that neonics, as currently used, are essential to farm production and farmer livelihood. There is significant potential in New York for more rational control of neonic insecticides to prevent further damage to the environment while benefitting producers and crop production.
- Experts worldwide are pointing out that our current path of food production is not sustainable and injurious to human health and the environment. Neonics, as currently marketed and used, offer a clear example of an unsustainable food production model.

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