Environmental Toxicology and Chemistry

Hazard/Risk Assessment

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Environ Toxicol Chem., **Accepted Article** • DOI: 10.1002/etc.3655

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This article includes online-only Supplemental Data.

Submitted 22 March 2016; Returned for Revision 11 September 2016; Accepted 16 October 2016

Abstract: A probabilistic ecological risk assessment (ERA) was conducted to determine the potential effects of acute and chronic exposure of aquatic invertebrate communities to imidacloprid arising from labeled agricultural and non-agricultural uses in the United States. Aquatic exposure estimates were derived using a higher tier refined modeling approach that accounts for realistic variability in environmental and agronomic factors. Toxicity was assessed using refined acute and chronic community-level effect metrics for aquatic invertebrates (i.e., species or taxon sensitivity distributions) developed using the best available data. Acute and chronic probabilistic risk estimates were derived by integrating the exposure distributions for different use patterns with the applicable species or taxon sensitivity distributions to generate risk curves, which plot cumulative probability of exceedance versus the magnitude of effect. Overall, the results of this assessment indicated that the aquatic invertebrate community is unlikely to be adversely affected by acute or chronic exposure to imidacloprid resulting from currently registered uses of imidacloprid in the United States.

Keywords: Probabilistic risk assessment, Neonicotinoids, Exposure modeling

1 INTRODUCTION

Imidacloprid is a first-generation neonicotinoid compound and one of the most widely used insecticides worldwide [1]. In 2009, imidacloprid accounted for over 40% of the neonicotinoid market in the United States [1]. Imidacloprid may be applied to either soil or foliage and is widely used in row crops (e.g., cotton, potatoes), greenhouse vegetables, forestry, vine crops, citrus, stone fruit and pome orchards, bush berries, and tree nuts to control a variety of pest insects, including aphids, Japanese beetles, lacebugs, leaf beetles, leafhoppers, leafminers, thrips, white flies, and others.

Although imidacloprid is not applied directly to water bodies, it may enter the aquatic environment through spray drift during application or storm water runoff shortly after application and has been detected in surface waters around the world [2]. In the aquatic environment, aquatic invertebrates have been shown to be more sensitive to imidacloprid than other aquatic taxa, due to the high specificity of imidacloprid and other neonicotinoids to the nicotinic acetylcholine receptors (nAChR) in the invertebrate central nervous system [3]. Therefore, the detection of imidacloprid in surface waters has raised concerns regarding potential direct negative impacts to aquatic invertebrate communities as well as indirect negative impacts to vertebrate species such as birds and fish that rely on aquatic invertebrates as prey [2-4].

The objective of this paper was to investigate the potential effects of acute and chronic exposure of aquatic invertebrate communities to imidacloprid arising from labeled agricultural and non-agricultural uses in the United States. This required a thorough review and evaluation of the available aquatic invertebrate toxicity data for imidacloprid, with a special focus on higher tier (mesocosm, semi-field and field) studies that considered more realistic exposure conditions. Additionally, exposure modeling was performed using a higher tier refined approach that accounted for variability in environmental and agronomic factors such as pesticide application date, weather, soils and slope of use sites, use site proximity to water bodies, and percent cropped area of targeted watersheds to develop realistic yet conservative estimates of aquatic exposure concentrations. This refined exposure modeling relied on the best available national scale datasets for agricultural and non-agricultural labeled uses of imidacloprid. Finally, risk was characterized using a probabilistic approach that considers both the probability of exposure to a range of pesticide concentrations and the magnitude of effects (if any) predicted to occur at those concentrations [5, 6].

2 METHODS

2.1 Identification of the best available data

A review of the scientific literature, registrant-sponsored studies, EPA's EcoTox database, existing water quality guideline documents (e.g., [7, 8]), and grey literature (e.g., [9]) was carried out to identify the best available data for the hazard portion of this assessment. All studies were evaluated using a transparent and reproducible assessment scheme developed based on data quality assessment factors described by NAS [5], EPA [10], Breton et al. [11], Klimisch et al. [12] and Hall et al. [13]. This assessment scheme is discussed in Knopper et al. [14] and the specific scoring rubrics for laboratory-based and higher tier (mesocosm, semi-field, and field) studies are described in the Supporting Information (Sections S1 and S2, respectively). On the basis of this evaluation, endpoints from each study were rated as acceptable, supplemental, or unacceptable for use in an ecological risk assessment (Supporting Information, Section S3). A study rating of acceptable indicates that the endpoint is ecologically relevant for potential population level effects, all essential information was reported, and the study was performed according to a complete and transparent study protocol that follows acceptable laboratory practices. Acceptable data are relevant and robust and are suitable for quantitative use in an ecological risk assessment. Data ranked as supplemental are also ecologically relevant, but their reliability is uncertain because their study conditions or methods either deviated from recognized protocol or were not reported in sufficient detail to enable evaluation. Therefore, supplemental data are not preferred for quantitative use in risk assessment, but may be considered qualitatively as part of a weight-of-evidence approach. Finally, data rated as unacceptable are not suitable for either quantitative or qualitative consideration in a risk assessment and should be excluded from further consideration.

2.2 Effects assessment

Frequently, measures of effect used in risk assessments are selected by compiling the available effects data and then choosing the most sensitive effects metric from the most sensitive test species to represent effects to the receptors of concern (e.g., [15]). This conservative approach is appropriate for screening-level assessments or when few data are available. However, for this refined assessment, toxicity data that were determined to be acceptable for quantitative use in an ecological risk assessment were available for a diverse group of aquatic invertebrate taxa. In addition, the purpose of this assessment was to assess risk to the aquatic invertebrate community rather than to specific sensitive species. Given the "functional redundancy" in aquatic invertebrate communities in the temperate zone [16, 17], multiple species are generally present as potential food sources for higher trophic level species and to perform critical functions such as cycling nutrients and organic matter. Hence the impairment of a sensitive species is not expected to alter overall ecosystem function because of other functionally similar species [18-21]. Additionally, populations of sensitive invertebrate species often recover quickly following exposure because of their high reproductive fecundity [22-26] and rapid recolonization from nearby refugia [27]. Therefore, alternative methods for deriving effects metrics were pursued that make use of all of the best available data to determine endpoints protective of the aquatic invertebrate community rather than relying on single most sensitive endpoints.

2.2.1 Acute effects

A species sensitivity distribution (SSD) was derived using the acute toxicity data that had an acceptable study rating (see Supporting Information Section S4). To derive the dataset for the SSD, only the most sensitive ecologically relevant endpoint for each species from each study was used. If more than one study reported suitable endpoints for the same species, the geometric mean value was calculated using the lowest acceptable endpoint values from each study. The final condensed dataset for imidacloprid is shown in Table 1.

SSD models were fit to the data in Table 1 using SSD Master v3 [28], an Excel-based tool that fits four non-linear regression models in log space (normal, logistic, extreme value, and Gumbel) to determine the best-fitting cumulative distribution function (CDF). Initially, SSD models were generated that included the acute *Daphnia magna* endpoint of 131,000 µg a.i./L (Table 1). However, this endpoint is three orders of magnitude higher than the next highest endpoint and removing the *Daphnia magna* endpoint improved the fit of the SSD models. A similar observation was made by RIVM [29]. Therefore, the *Daphnia magna* endpoint was excluded from the final SSD analysis.

Of the SSDs generated by SSD Master v3 for acute laboratory-based toxicity data (Table 1) with *Daphnia magna* excluded, the best-fitting SSD was provided by the logistic model in log space based on the results of the Anderson-Darling (AD) goodness-of-fit test statistic (AD=0.26, $p>0.05$), mean square error (0.0034) and mean square error lower tail (0.0168). Visual inspection of the fitted CDFs also indicated that the logistic model closely followed the data (Figure 1). The logistic model is described by Equation 1.

$$f(x) = \frac{1}{1 + e^{-\frac{x-\mu}{s}}} \quad \textbf{(1)}$$

where *x* is concentration (in µg a.i./L), and *f(x)* is the proportion of taxa affected. The fitted location and scale parameters, μ and *s*, for this model were 1.44 and 0.409, respectively. The predicted 5% hazard concentration (HC_5) for this SSD was 1.73 µg a.i./L (95% CI from 1.01 to 2.97 µg a.i./L).

2.2.2 Chronic effects

The literature review and study evaluation process identified two distinct sets of chronic effects data for imidacloprid. The first dataset included the results of chronic toxicity tests conducted in laboratories according to standardized protocols that are well established, accepted, and reproducible. These laboratory-based studies generally tested single species exposed to constant concentrations of imidacloprid maintained by static renewal (e.g., Roessink et al., [30]) or flow-through test systems (e.g., Ward [31]). The second dataset included the results of chronic mesocosm, semi-field and field studies designed to more closely mimic natural conditions and exposure from registered used patterns. These studies considered factors such as varied exposure concentrations over time, the presence of sediment in the test system, the potential for natural recolonization and recovery, the effects of natural lighting and weather fluctuations, and/or the impacts of interactions between multiple species. Separate chronic effects metrics were developed using each of these two datasets.

Standard laboratory-based chronic effects

An SSD was derived using the chronic laboratory-based toxicity data that had an acceptable study rating (see Supporting Information section S5). The approach taken to derive the chronic SSD using laboratory-based data was the same as described above for the acute SSD. The final condensed chronic dataset is shown in Table 2.

SSD models were fit to the chronic effects metrics presented in Table 2 using SSD Master v3. As with the acute SSD, the *Daphnia magna* endpoint was excluded from chronic SSD analysis because the species is two orders of magnitude more tolerant than then the next most tolerant species.

Of the SSDs generated by SSD Master v3 for chronic laboratory based toxicity data (Table 2) with the *Daphnia magna* endpoint excluded, the best-fitting SSD was provided by the logistic model in log space. This model had the best fit based on the results of the Anderson-

Darling (AD) goodness- of-fit test statistic (AD=0.226, $p>0.05$), mean square error (0.0035) and mean square error lower tail (0.017). Visual inspection also indicated an excellent fit to the data (Figure 2). This model is described by Equation 1. The fitted location and scale parameters, μ and s, for this logistic model were 3.15 and 0.529, respectively. The HC5 was 0.039 µg a.i./L (95% CI from 0.02 to 0.075 µg a.i./L).

Chronic effects derived from chronic mesocosm, semi-field and field ('higher tier') studies

The higher tier toxicity data (i.e., NOECs for density, abundance, emergence, and mortality) that were judged to be acceptable are listed in full in the Supporting Information (Section S6). A comparison of these data with the chronic laboratory-based toxicity data reported in Table 2 reveals the degree to which the incorporation of more environmentally realistic conditions and species interactions within the higher tier studies can affect the apparent sensitivity of aquatic invertebrates to imidacloprid. For example, Roessink et al. [30] reported a chronic (28-day) LC10 of 0.041 µg a.i./L for *Cloeon dipterum* under standard laboratory conditions (Table 2), whereas chronic NOECs for this species reported from the higher-tier studies ranged from 0.243 to 9.4 µg a.i./L (Supporting Information Section S6, [25, 32, 33]). These results suggest that standard chronic laboratory test may significantly overestimate the sensitivity of aquatic invertebrates to imidacloprid under realistic field conditions. Therefore, significant consideration should be given to the higher tier study data when making risk conclusions.

In many of the higher tier studies, toxicity endpoints were reported for taxonomic groupings at a higher level of organization than species (e.g., at the genus, tribe, subfamily, or order level) (see Supporting Information Section S6). As such, we could not derive an SSD using these data. However, given the wealth of acceptable data available, we were able to derive a sensitivity distribution at a higher level of taxonomic characterization. Therefore, the chronic higher-tier NOECs were grouped at the level of family, subfamily, or class to generate a Taxon Sensitivity Distribution (TSD, see data summary in Table 3).

To select the endpoints used to derive the TSD, the most sensitive acceptable NOEC for each taxonomic group from each study was compiled. Following the approach used for the acute and chronic laboratory-based SSDs, the geometric mean was used to summarize multiple endpoints reported for the same family, subfamily, or class from different studies (Table 3). In some cases, this approach excluded less sensitive NOECs from studies where recovery was taken into account. For example, in Moring et al. [22], the test system was observed for three months following cessation of treatment. During the initial exposure period, amphipods, copepods, and macroinvertebrates experienced declines in abundance, resulting in NOECs as low as 2 µg a.i./L (Table 3). However, full recovery of all species was observed within eight weeks of the final treatment. Therefore, Moring et al. [22] suggested that the next highest treatment concentration (6 µg a.i./L) should be adopted as a regulatory NOEC for these families. Likewise, Ratte and Memmert [25] reported NOECs as low as 0.6 µg a.i./L (Table 3) when recovery was not considered, but also reported "no observed ecologically adverse effect concentrations" (NOEAECs) ranging from 9.4 µg a.i./L to 23.5 µg a.i./L based on the complete recovery of emerging insects and zooplankton within eight weeks of the last application. Although these recovery-based endpoints were excluded from the TSD dataset (Table 3), they suggest that risk to aquatic invertebrates from chronic imidacloprid exposure may be overestimated when the potential for recovery is not accounted for.

Additionally, the NOECs reported in the higher tier studies were based on initial treatment concentrations following the first application (either nominal concentrations that were confirmed analytically or measured concentrations). However, the higher tier studies did not seek to maintain a constant concentration over the course of the study period. The pulse exposure regimes used in these studies varied with respect to number of applications (1 to 4 applications) and retreatment interval (7, 14, or 21 d intervals) in order to mimic potential drift or runoff events. Following application, the imidacloprid concentration declined due to the natural degradation and dissipation of imidacloprid in aquatic systems. Therefore, to develop an effects metric that could be compared to a constant chronic exposure duration, time-weighted average concentration estimates were determined for the reported NOECs from the day of the first application to 21 days following the final application using the degradation half-life (DT50) of 11.6 days reported by Roessink et al. [32] and assuming first-order elimination kinetics per Equation 2. The 21 day interval was selected as this corresponded to the most common application interval in the higher tier studies with multiple applications. Additionally, a consistent cutoff was required to ensure that exposure estimates were not severely underestimated in studies that had very long durations.

$$\text{for } i = 1 \text{ to n}, \qquad C_i = \frac{C_{i-1}(1 - e^{-kt})}{kt} \qquad (2)$$

Where n = The number of treatments applied in the study; C_i = The time-weighted average concentration (*μg a.i./L)* from application *i* to the next application or to 21 days following application *i* if there were no further applications. Note, C_0 represents the initial concentration immediately following the first treatment as reported in the study. t = the duration of time over which the time weighted average was calculated (i.e., the retreatment interval for *i* = 1 to n-1 and 21 days for *i*=*n*. k = ln(2)/DT50

For studies with only one treatment application, the time-weighted average concentration from the day of application to 21 days after application was calculated using Equation 2 directly. For studies with more than one treatment application, Equation 2 was applied to calculate time-weighted average concentrations between each application and from the final application to 21 days after that application and then an overall time-weighted average for the whole study was estimated using Equation 3.

$$\mathrm{TWA} = \frac{(\sum_{i=1}^{n-1} C_i * t) + C_n * 21}{(n-1)t + 21} \tag{3}$$

Where

TWA = the overall study length time-weighted average concentration (µg a.i./L)

C_i = the time-weighted average concentration (µg a.i./L) from the *i*th application to the next application if there was a subsequent application or from the *i*th application to 21 days following that application if there were no further treatments (see Equation 2).

n = the number of treatments applied in the study

t = the length of the retreatment interval

The resulting time-weighted average NOEC estimates are reported in Table 3.

Although it is not possible to conclude that effects observed in the higher tier studies were the result of the longer term, lower exposure concentration represented by the TWA rather than due to an acute peak exposure or repeated peak exposures, using the TWA results in lower, more conservative NOEC values than those originally reported in the studies (see Table 3). Therefore, since the TWA approach is useful for standardizing results between different studies and exposure regimes while potentially increasing the conservatism of the assessment, it was considered appropriate and allowed for the use of all relevant studies in the assessment.

Taxa-specific sensitivity distributions (TSDs) were fit to the time-weighted average NOEC estimates in Table 3 using SSD Master v3. The best-fitting distribution was the Gumbel model (in log space) based on the results of the Anderson-Darling (AD) goodness-of-fit test statistic (AD=0.612, p>0.05), mean square error (0.0061) and mean square error lower tail (0.055). The Gumbel model closely followed the data (Figure 3). This model is described by Equation 4:

$$f(x) = e^{-e^{\frac{(\mu - x)}{b}}} \quad \textbf{(4)}$$

where *x* is concentration (in µg a.i./L), and *f(x)* is the proportion of taxa affected. The fitted location and scale parameters, *µ* and *s*, for this model were 3.38 and 0.347, respectively. The predicted 5% hazard concentration (HC_5) for this TSD was 1.01 µg a.i./L (95% CI from 0.692 to 1.47 µg a.i./L).

2.3 Exposure assessment

The objective of the refined aquatic exposure modeling was to derive comprehensive and realistic distributions of aquatic estimated environmental concentrations (EECs) associated with representative imidacloprid use patterns. The approaches used to select use patterns and derive refined aquatic exposure estimates for agricultural and non-agricultural uses are described below.

2.3.1 Agricultural use patterns

An evaluation of all labeled agricultural uses of imidacloprid was not practical. Therefore, a subset of crop use patterns was selected to represent various crop types and geographic regions and to include higher vulnerability uses (i.e., uses predicted to have the highest EECs using standard Tier II aquatic exposure modeling, see Supporting Information Section S7). The selected crop use patterns (Table 4) included major crops (e.g., soybeans and potatoes) and minor crops (e.g., fruiting vegetables and leafy greens) as well as both row crops and orchard crops (citrus). Additionally, high vulnerability uses such as cucurbits were selected (see Supporting Information Section S7). Overall, the crop use patterns chosen for refinements represent a broad spectrum of the imidacloprid agricultural uses and are expected to provide a comprehensive understanding of the likely exposure associated with registered agricultural uses of imidacloprid.

Exposure modeling for the agricultural use patterns incorporated EPA's standard aquatic exposure modeling tool, the Surface Water Concentration Calculator (SWCC) model version 1.106 [34], and the standard assumptions of a 10 hectare field draining into a 1 hectare, 2 meter deep farm pond. In addition, a specialized vegetative filter strip model (VFSMOD version 4.2.4) [35] was used to simulate the mitigating effects of the 10 foot vegetative filter strip (VFS) that is required on the imidacloprid agricultural label.

For all agricultural scenarios except for the two potato scenarios, Latin Hypercube sampling of the probability distributions of key characteristics (i.e., application date(s), weather stations, soil profile and land surface slope, pond-integrated spray drift fraction and percent cropped area) was used to develop 1,000 unique sets of the required model input parameters. This was done for each use scenario rather than relying on standard EPA screening level scenarios representative of high runoff and erosion potential to define model input values. Using these sets of parameters, 1,000 runs of 30-year simulations were modeled for each use pattern, resulting in a distribution of 30,000 annual maximum EECs for each exposure duration of interest (i.e., peak, 48-hour, 96-hour, 21-day, 60-day, and 90-day average periods).

For the potato scenarios, a different approach was applied that did not include the probabilistic characterization of the model inputs or accounting for a VFS using VFSMOD. Rather, refinements to the potato exposure modeling focused on parameterizing the application method to reflect in-furrow application at planting. This application method limits potential imidacloprid runoff to surface water exposure as the majority of the pesticide is applied at the planting depth and below the active runoff zone in the soil.

Further details regarding the models used as well as identification and development of the required model input values for the refined agricultural use exposure modeling are provided in Supporting Information, Section S8.

2.3.2 Non-agricultural use patterns

For the non-agricultural use patterns, the SWCC model (version 1.106, [34]) was also used to develop distributions of refined aquatic exposure estimates. Exposure refinements were achieved by simulating more realistic model inputs with respect to application timing, runoff vulnerability, off-target overspray, percent cropped area (PCA) (for golf course uses), and percent treated area (PTA). The specific model input refinements varied across the three types of non-agricultural use patterns assessed (i.e., golf course turf, residential, and nursery uses). These refinements are described in detail in Supporting Information, Section S9. Overall, these refinements produced distributions of annual maximum EECs with 30,000 values (i.e., 30 modeled years x 1,000 customized model input scenarios) for the golf course turf use patterns, 10,740 or 10,950 values for the residential use patterns (30 years modeled based on either 358 or 365 initial application dates), and between 19,890 and 24,750 values for the nursery use patterns (30 years modeled for each of 3 different runoff curve number assumptions for between 221 and 275 possible initial application dates).

2.4Risk characterization

For the refined risk assessment, risk curves were generated by integrating the distributions of refined exposure estimates with the refined measures of effect. Risk curves plot the cumulative probability of exceedance versus the magnitude of effect (i.e., the probability of exceeding effects of differing magnitude). The use of such risk curves, also known as “joint probability curves” [6], in risk analysis has been recommended by both the NAS [5] and the EPA [36-38]. For each risk curve, the area under the curve (AUC) was calculated. Expressed as a percentage, AUC can range from 0% to 100%, with a lower AUC representing a lower overall probability and magnitude of risk. Although the AUC does not provide information about the shape of the risk curve, it can be used to summarize and compare risk scenarios. For this assessment, the AUC of each risk curve was used to categorize risk qualitatively as *de minimis*, low, intermediate, or high for each use pattern as follows.

- If the AUC was less than the AUC associated with the curve produced by risk products (risk product = exceedance probability x magnitude of effect) of 0.25% (e.g., 5% exceedance probability of 5% or greater effect = 0.25%), then the risk was categorized as *de minimis*. The AUC for risk products of 0.25% is 1.75%;
- If the AUC was equal to or greater than 1.75%, but less than 9.82% (i.e., the AUC for a curve with a constant risk product of 2%), then the risk was categorized as low;
- If the AUC was equal to or greater than 9.82%, but less than 33% (i.e, the AUC for a curve with a constant risk product of 10%), then the risk was categorized as intermediate; and
- If the AUC was equal to or greater than 33%, then the risk was categorized as high.

The risk category boundaries, as described above, are shown in Figure 4. These risk categories are intended to be summary descriptors of the risks to aquatic invertebrates exposed to imidacloprid near treated areas. Similar risk categorization schemes have previously been applied to ecological risk assessments for other pesticides [39-41] and contaminated sites [42]. Overall, these risk boundaries are designed to be protective of the aquatic invertebrate community and therefore do not focus on effects to single species (or taxa). The high reproductive potential of most aquatic ecosystems enables them to rebound in a relatively short time after experiencing low to intermediate adverse effects [23, 24, 26, 43]. However, Liess and Schulz [44] showed that recovery may take longer (months to years) for aquatic invertebrates when local populations are extirpated. Thus, exposure scenarios in the high risk category would be of concern due to the long periods of time potentially required for population recovery.

RESULTS AND DISCUSSION

Exposure modeling results

Agricultural use patterns. A summary of the refined 90th percentile estimated environmental concentrations (EECs) at different exposure durations is provided in Table 5. Mississippi soybean had the highest 90th percentile concentrations, ranging from 0.30 µg a.i./L (annual maximum) to 0.03 µg a.i./L (annual average). The higher Mississippi soybean concentrations relative to the other crop scenarios were driven by higher spray drift potential resulting from the aerial application method and close proximity of treated areas to receiving water, as well as higher percent cropped areas (PCAs) around water bodies relative to some other crops. The California and Florida citrus scenarios followed soybean with 90th percentile concentrations ranging from 0.14 µg a.i./L (Florida, annual maximum) to 0.01 µg a.i./L (annual average). Concentration predictions for California and Florida citrus were impacted by the higher PCA distributions for those crop scenarios. Overall, the highest maximum concentrations for each scenario were driven by environments with high runoff potential, high drift and high PCA. The maximum concentrations for each scenario have a very low probability (1/30,000) of occurring and represent extreme worst-case conditions given that the corresponding 90th percentile concentrations are orders of magnitude lower than the maxima.

Scenarios with the soil chemigation/drench application method (no spray drift) resulted in lower concentrations than the scenarios with ground and aerial applications. Seed treatment application of imidacloprid has no potential for drift and represents much lower vulnerability use patterns than the use patterns evaluated in this refined assessment. Soil application results in lower surface water concentrations for all crops compared to foliar application of imidacloprid. Of the scenarios with soil application, California tomato had the highest refined 90th percentile annual maximum concentrations because it had some of the highest PCAs and the lowest median vegetative filter efficiency (41.7%, Supporting Information Section S8).

2.4.1 Non-agricultural use patterns

Summaries of the refined 90th percentile estimated environmental concentrations (EECs) at different exposure durations for golf course turf, residential, and nursery use patterns are provided in Table 6, Table 7, and Table 8, respectively. For each use pattern, EECs are shown both with and without the application of a percent treated area (PTA) refinement. As described in Supporting Information Section S9, the EECs derived for the 100% PTA condition assume that imidacloprid is applied to all possible use sites for the applicable use pattern. However, it was possible to refine the PTA estimates for each use pattern by combining various geographic datasets with imidacloprid sales data from 2013 and 2014 for golf course, residential, and nursery use products (see description in Supporting Information Section S9). Using these refined PTAs in the modeling produced EECs that reflect more realistic exposure conditions.

2.4.2 Comparison to surface water monitoring results

To provide context for the refined exposure modeling results, modeled concentrations were compared to available surface water monitoring data for imidacloprid from the EPA Storage and Retrieval (STORET) Data Warehouse [45], US Department of Agriculture (USDA) Pesticide Data Program (PDP) [46], US Geological Survey (USGS) National Water Quality Assessment (NAWQA) Program [47], and the USGS National Water Information System (NWIS) [48, 49]. More than a decade of water sample data were available from some monitoring stations in these federal monitoring programs. For example, in the USGS NAWQA program, water quality was monitored in 42 study units located across the US between 1991 and 2012. These study units were designed to “cover a variety of important hydrologic and ecological resources, critical sources of contaminants, including agriculture, urban and natural sources; and a high percentage of population served by municipal water supply and irrigated agriculture” [50]. Furthermore, the NAWQA study units are representative of a wide range of agricultural practices (e.g., crops, tillage, irrigation, drainage and chemical use) and landscapes (e.g., geology, soil type, topography, climate and hydrology) [51].

In these federal monitoring datasets, imidacloprid was detected in 4 to 18.5% of all samples, with detection limits ranging from 0.0015 to 0.5 μg a.i./L. Conservative estimates of 90th percentile imidacloprid concentrations in these datasets; calculated by setting all non-detect samples to the reported limit of detection, limit of quantitation, or reporting limit; ranged from 0.02 to 0.106 μg a.i./L (Supporting Information Section S10). Additionally, only 0.23% of these monitoring samples exceeded 1 μg a.i./L. In comparison, the refined exposure modeling in this assessment predicted 90th percentile annual maximum peak EECs ranging from 0.006 to 0.304 μg a.i./L for agricultural use patterns, from 0.18 μg a.i./L to 4.42 μg a.i./L for non-agricultural use patterns assuming 100% PTA, and from 0.0026 μg a.i./L to 0.061 μg a.i./L for non-agricultural use patterns with PTA refinements applied. Although a direct comparison between modeling data and monitoring data is not possible due to differences in the frequency and duration of data collection, the ranges of modeled EECs are similar to or exceed the ranges of values reported in the monitoring datasets, suggesting that the refined exposure assessment are generally realistic with a bias to being moderately conservative.

Risk characterization

Acute risk. Acute risk curves were generated by integrating the probabilistic distribution of the maximum peak EECs from each modeled year for each customized model input scenario with the acute lab-based SSD shown in Figure 1. Using the AUC statistic, risk to aquatic invertebrates from acute exposure to imidacloprid was categorized as *de minimis* for all agricultural use patterns (Table 9). For example, consider the risk curve for Mississippi Soybean (MS soybean), which had the highest AUC of the agricultural use patterns (0.342%, Table 9). This risk curve lies very close to the x- and y-axes, indicating very low probabilities of adverse effects (mortality) to the most sensitive species and negligible probabilities of effects to less sensitive species (Figure 5).

For the non-agricultural use patterns when 100% PTA was assumed, eight of 12 use patterns were categorized as posing *de minimis* risk (Table 10). The remaining four use patterns, which were categorized as posing a low risk, were nursery uses in Florida, Michigan, New Jersey, and Tennessee (Table 10). However, nursery use of imidacloprid is typically a spot treatment. The assumption of 100% PTA overestimates exposure and therefore risk. With PTA refinements (described in detail in Supporting Information Section S9), nursery uses in these states were categorized as being at *de minimis* risk (Table 10). For additional context, the acute risk curve for Golf Course Turf (Pennsylvania Turf, Refined PTA) is shown in Figure 6. This risk curve represents the worst case scenario (highest AUC) for acute risk due to non-agricultural uses for the refined PTA assumption. This worst-case risk curve lies very close to the x- and y-axes, indicating negligible probabilities of exceeding the effects metric used in the acute lab-based SSD (i.e., acute LC50 or EC50 (immobilization)) for any significant proportion of species (e.g., the probability of exceeding the effects metric for 5% of species is 0.00333%) (Figure 6). Overall, no unreasonable risk of adverse effects to aquatic invertebrates is expected from acute exposure to imidacloprid from agricultural or non-agricultural uses.

2.4.3 Chronic risk

Two sets of chronic risk curves were generated to characterize chronic risk. In the first set, risk curves were generated by integrating the distribution of the maximum 21-day average EECs from each modeled year with the chronic lab-based SSD. However, as noted in the chronic effects assessment, the chronic higher tier TSD is considered the best available data for chronic risk assessment as it is more directly relatable to community level effects and the studies used to generate the TSD were conducted under more realistic environmental conditions. Therefore, the second set of chronic risk curves integrated the same chronic exposure data with the chronic higher tier TSD (i.e., a taxon-specific sensitivity distribution composed of chronic higher-tier NOECs established at the family/sub-family taxonomic level).

For the agricultural use patterns with risk curves generated using the chronic lab-based SSD, seven out of 14 were categorized as ‘low’ risk, while the remaining were *de minimis* risk (Table 11). In contrast, chronic risk curves generated using the best available data (i.e., the chronic higher tier TSD) were categorized as *de minimis* for all 12 modeled agricultural use patterns (see Chronic II risk curves, Table 11). The worst-case (highest AUC) agricultural use risk curve generated using the chronic higher tier TSD was Florida Citrus (Florida Citrus, Figure 7), with an AUC of 0.0734% (Table 11). This risk curve shows an extremely low probability of exceeding the TSD effects metric for even a small percentage of taxa (Figure 7).

For the non-agricultural use patterns, the risk curves derived using the chronic lab-based SSD and EECs developed assuming 100% PTA indicated low risk for one use pattern, intermediate risk for eight use patterns and high risk for three use patterns (Table 12). However, when the PTA refinement was applied to the EECs for these non-agricultural use patterns, the risk curves generated using the chronic lab-based SSD indicated that risk was either *de minimis* (seven use patterns) or low (five use patterns), with the worst-case (highest AUC of 2.66%) risk curve for Golf Course Turf (Pennsylvania Turf) (Table 12). For risk curves derived using the chronic higher tier TSD, risk was categorized as *de minimis* for the majority of non-agricultural uses, regardless of whether the PTA refinement was applied (Table 12). Additionally, with the PTA refinement, all non-agricultural use patterns had *de minimis* chronic risk (Table 12). Of these, the worst-case scenario (highest AUC of 0.00160%, Table 12) was observed for Golf Course Turf (Florida Turf, Figure 8). This risk curve exhibits negligible probabilities of exceedance of the effects metric used in the chronic higher tier TSD (e.g., probability of exceeding the higher tier family/subfamily based NOECs for the 5% most sensitive taxa is 0.00667%, Figure 8). Given that the chronic higher tier TSD and the PTA-refined EECs represent the best available data, no unreasonable risk of adverse effects to aquatic invertebrates is expected from chronic exposure to imidacloprid from non-agricultural uses.

2.5 Assumptions and uncertainties

This assessment was developed with the best available data and information gained from long-term registered imidacloprid use in the United States. However, uncertainty is inherent in every assessment. The probabilistic exposure modeling approach accounted for several sources of uncertainty associated with imidacloprid use sites and receiving waters by sampling distributions of application timing, weather, soil characteristics, PCA, and drift fraction reflective of crop proximity relative to pond locations and pond surface area. This approach reduced the uncertainty in the model assumptions, inputs, and predicted imidacloprid concentrations by incorporating best available datasets to better characterize the range of conditions leading to potential imidacloprid exposure. In accounting for the uncertainty in the environmental and agronomic conditions assessed, the range of potential imidacloprid exposure concentrations increased, because a much broader set of conditions were considered.

Additionally, the exposure modeling maintained several conservative assumptions that may have overestimated imidacloprid EECs. For example, the EPA standard farm pond considered in the modeling does not account for potential outflow during large runoff events or for the addition of eroded sediment into the water body. Accounting for outflow would lead to lower EECs, as some chemical would be flushed downstream rather than being held in the static water body. A higher sediment concentration in the water body could lead to a larger portion of the pesticide in the water column binding to sediment, making it non-bioavailable. Other conservative assumptions included use of the maximum label application rate for non-agricultural scenarios although typical applications are often at lower rates, assuming that 100% of the crop area was treated for agricultural scenarios even though imidacloprid does not have 100% of the market share, ignoring agricultural practices that may reduce pesticide transport off-field (e.g., conservation tillage and avoiding pesticide application on rainy days), and assuming that all pesticide on foliage at harvest time will be returned to the soil although harvested material may be removed from the field.

With respect to the measures of ecological effects applied in this assessment, a thorough literature review and study evaluation process was carried out to identify the best available data. Because only high quality data were considered, a high level of confidence can be placed on the data selected for use in this assessment. Additionally, acceptable toxicity data were identified for a much more diverse group of aquatic invertebrates than the core EPA pesticide registration data requirements (i.e., one acute freshwater invertebrate (EC50) study, one freshwater invertebrate life cycle (NOEC) study, and one acute estuarine invertebrate (LC50/EC50) per 40 CFR 40, §158.630 (2015)). The inclusion of data for a larger variety of aquatic invertebrates also helps to reduce uncertainty regarding the effects assessment. Finally, the effects assessment did not consider the potential for recovery of affected species through recolonization, high reproductive potential, or other means and was thus conservative.

CONCLUSIONS

Aquatic invertebrates are highly sensitive to imidacloprid exposure under certain conditions. For standard laboratory-based toxicity tests that received an acceptable rating in the study evaluation, acute LC50 values were as low as 2.07 µg a.i./L (Table 1) and chronic NOEC values were as low as 0.041 µg a.i./L (Table 2). The chronic NOECs observed in the higher-tier studies were considerably less sensitive, but still reflective of potentially environmentally relevant aquatic exposure concentrations. For example, the most sensitive acceptable chronic NOEC value used to develop the chronic higher-tier TSD was 0.243 µg a.i./L (Table 3) while the 90th percentile imidacloprid concentrations reported in federal monitoring datasets in recent years ranged from 0.02 to 0.106 µg a.i./L (Supporting Information Section S10). However, a direct comparison of monitoring data to effects data overlooks issues such as frequency and duration of exposure, geographic variability, and the potential for changes in use patterns or label instructions to alter exposure over time. Therefore, aquatic exposure modeling was used in this assessment to derive comprehensive and realistic exceedance probabilities of imidacloprid aquatic concentrations for various ecologically relevant durations. It was then possible to generate risk curves that plot cumulative probability of exceedance versus the magnitude of effect by integrating the modeled exposure distributions with relevant SSDs and TSDs derived using the best available data. The results of this probabilistic risk assessment found that based on the best available data (i.e., refined exposure modeling and higher-tier toxicity data), aquatic invertebrate communities are not likely to be at risk from acute or chronic exposure to imidacloprid from registered uses in the United States.

Supplemental Data—The Supplemental Data are available on the Wiley Online Library at DOI: 10.1002/etc.xxxx.

Acknowledgment—This research was supported by Bayer CropScience. We would also like to acknowledge the contributions of R. Breton for his contribution with respect to the design of the study evaluation rubrics and G. Manning for assisting in the study evaluations and compiling the surface water monitoring data.

Data Availability—For additional data, please see the supporting information file.

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Figure 1. Acute lab-based SSD constructed using acceptable laboratory-based acute toxicity endpoints for aquatic invertebrates (i.e., acute LC50 or EC50 (immobilization) values).

Figure 2. Chronic lab-based SSD constructed using acceptable laboratory-based chronic toxicity endpoints for aquatic invertebrates (NOEC, EC10, or LC10 values representative of ecologically relevant effects to growth, mortality, or reproduction).

Figure 3. Chronic higher-tier TSD constructed using estimated time weighted average NOECs from higher tier toxicity studies.

Figure 4. Risk curves defining the AUC boundaries for risk categorization.

Figure 5. Acute risk curve for Mississippi soybean.

Figure 6. Acute risk curve for Golf Course (Pennsylvania Turf) with percent treated area (PTA) refinement.

Figure 7. Chronic risk curve for Florida Citrus using chronic higher tier TSD.

Figure 8. Chronic risk curve for Golf Course (Florida Turf) with percent treated area (PTA) refinement using higher tier TSD.

Table 1. Acceptable laboratory-based acute toxicity endpoints (LC50 or EC50 (immobilization) values) for aquatic invertebrates used to derive acute SSD

Scientific Name (Duration)	***LC50 or EC50 (immobilization) (µg a.i./L)***	***Reference***
Ceriodaphnia dubia (48 h)	2.07	Chen et al. [52]
Caenis horaria (96 h)	6.68	Roessink et al. [30]
Simulium vittatum (48 h)	6.75	Overmyer et al. [53]
Chironomus tentans (96 h)	10.5	Gagliano [54]
Cloeon dipterum (96 h)	26.3	Roessink et al. [30]
Mysidopsis bahia (72 – 96 h)	34.8[a]	Lintott [55]; Ward [56]
Plea minutissima (96 h)	37.5	Roessink et al. [30]
Hyalella azteca (96 h)	55	England and Bucksath [57]
Chironomus riparius (24 h)	55.2	Dorgerloh and Sommer [58]
Chaoborus obscuripes (96 h)	294	Roessink et al. [30]
Asellus aquaticus (96 h)	316	Roessink et al. [30]
Daphnia magna (48 h)	131,000[b]	Riebschläger [59]

a. Geometric mean of 33.7 µg a.i./L [56] and 36 µg a.i./L [55].

[b] Not included in final SSD due to its relative insensitivity. Removing this endpoint improved overall fit of SSD.

Table 2 Acceptable laboratory-based chronic toxicity endpoints (NOEC, EC10, or LC10 values representative of ecologically relevant effects to growth, mortality, or reproduction) for aquatic invertebrates used to derive chronic laboratory-based SSD

Scientific Name	***Effect Concentration (µg a.i./L)***	***Reference***
Cloeon dipterum	0.041	Roessink et al. [30]
Mysidopsis bahia	0.163	Ward [31]
Caenis horaria	0.235	Roessink et al. [30]
Chironomus tentans	1.14	Stoughton et al. [60]
Asellus aquaticus	1.35	Roessink et al. [30]
Chironomus riparius	1.70[a]	Bruns [61, 62]; Dorgerloh [63-65]; Dorgerloh and Sommer [66-68]
Chaoborus obscuripes	1.99	Roessink et al. [30]
Plea minutissima	4.35	Roessink et al. [30]
Gammarus pulex	19.2[b]	Hendel [69], Roessink et al. [30]
Sialis lutaria	25.1	Roessink et al. [30]
Daphnia magna	1,236[c,d]	Heimbach [70]; Ieromina et al. [71]; Jemec et al. [72]; Pavlaki et al. [73]; Young and Blakemore [74]

a. Geometric mean of 0.56 µg a.i./L [62], 1.33 µg a.i./L [61], 1.8 µg a.i./L [63], 1.8 µg a.i./L [64], 1.87 µg a.i./L [68], 2.09 µg a.i./L [66], 2.28 µg a.i./L [67], 3.19 µg a.i./L [65]

b. Geometric mean of 5.77 µg a.i./L [30], 64 µg a.i./L [69]

c. Geometric mean of 320 µg a.i./L [70], 1250 µg a.i./L [72], 1,800 µg a.i./L [74], 2000 µg a.i./L [71], 2000 µg a.i./L [73]

[d]Not included in final SSD due to its relative insensitivity. Removing this endpoint improved overall fit of SSD

Table 3 Acceptable higher tier toxicity data for aquatic invertebrates (i.e., NOECs for density, abundance, emergence, and mortality) grouped by family, subfamily (Chironiminae, Orthocladinae, and Tanypodinae only), or class (Copepoda only) used to derive chronic higher-tier TSD

Scientific Name	***Effect Concentration Reported in Study (μg a.i./L)*** [a]	***Time-weighted Average NOEC Estimates (μg a.i./L)***	***Reference***
Baetidae	0.816[b]	0.581	Roessink et al. [32], Ratte and Memmert [25], Roessink and Hartgers [33], Moring et al. [22]
Chironominae	1.90[c]	1.48	Ratte and Memmert, [25], Moring et al., [22]
Hydrophilidae	2.00	1.87	Moring et al. [22]
Caenidae	2.00	1.87	Moring et al. [22]
Hydroptilidae	2.00	1.87	Moring et al. [22]
Naididae	3.80	2.47	Ratte and Memmert [25]
Chaoboridae	3.80	2.47	Ratte and Memmert [25]
Orthocladiinae	3.80	2.47	Ratte and Memmert, [25]
Copepoda	7.51[d]	5.85	Moring et al. [22], Ratte and Memmert [25]
Glossiphoniidae	9.40	6.12	Ratte and Memmert [25]
Daphniidae	9.40	6.12	Ratte and Memmert [25]
Planorbidae	9.40	6.12	Ratte and Memmert [25]
Tipulidae	12.0	6.84	Kreutzweiser et al. [75]
Tanypodinae	13.7[e]	10.7	Ratte and Memmert [25], Moring et al. [22]
Pteronarcyidae	24[f]	13.7	Kreutzweiser et al. [75], Kreutzweiser et al. [76]

a. Endpoints selected for this assessment represent the lowest endpoint for each taxa from a single study or a geometric mean of the lowest values from multiple studies. A conservative approach was taken for selecting the endpoints that did not account for recovery.
b. Geometric mean of 0.243 μg a.i./L [32], 0.6 μg a.i./L [25], 1.52 μg a.i./L [33], and 2 μg a.i./L [22].
c. Geometric mean of 0.6 μg a.i./L [25]and 6 μg a.i./L [22].
d. Geometric mean of 6 μg a.i./L [22] and 9.4 μg a.i./L [25].
e. Geometric mean of 9.4 μg a.i./L μg a.i./L [25]and 20 μg a.i./L [22].
[f] Geometric mean of 12 μg a.i./L [75] and 48 μg a.i./L [76]

Table 4 Imidacloprid crop use patterns for refined exposure assessment and the individual crop scenarios included in association with each use pattern

Crop Use Pattern	***US EPA Standard Crop Scenarios Assessed***
Cucurbit	FLcucumberSTD, MImelonSTD, NJmelonSTD, MOmelonSTD
Citrus	FLcitrusSTD, CAcitrus_WirrigSTD
Leafy greens	CAlettuceSTD
Fruiting vegetables	FLpeppersSTD, FLtomatoSTD_V2, PAtomatoSTD, CAtomato_WirrigSTD
Brassica	FLcabbageSTD
Potato	IDNpotato_WirrigSTD, MEpotatoSTD
Soybean	MSsoybeanSTD

Table 5 Summary of refined 90th percentile exposure predictions for agricultural use patterns (µg a.i./L)

Crop Scenario[a]	***Crop Use Pattern***	***Annual Maximum***	***48-hr Average***	***96-hr Average***	***21-day Average***	***60-day Average***	***90-day Average***	***Annual Average***
Florida citrus	Citrus, foliar airblast	0.14	0.14	0.13	0.10	0.06	0.05	0.01
California citrus	Citrus, foliar airblast	0.09	0.09	0.08	0.06	0.04	0.03	0.01
California lettuce	Leafy greens, foliar ground-boom	0.07	0.07	0.06	0.05	0.03	0.03	0.01
Florida cabbage	Brassica, foliar ground-boom	0.006	0.006	0.006	0.004	0.003	0.002	0.001
Mississippi soybean	Soybean, foliar aerial	0.30	0.29	0.28	0.21	0.13	0.10	0.03
Florida peppers	Fruiting vegetables, soil chemigation/drench	0.025	0.024	0.022	0.016	0.009	0.007	0.002
Florida tomato	Fruiting vegetables, soil chemigation/drench	0.013	0.013	0.012	0.009	0.005	0.004	0.001
Pennsylvania tomato	Fruiting vegetables, soil chemigation/drench	0.006	0.006	0.005	0.004	0.002	0.002	0.0005
California tomato	Fruiting vegetables, soil chemigation/drench	0.15	0.15	0.14	0.11	0.06	0.04	0.01
Florida cucumber	Cucurbits, soil chemigation/drench	0.021	0.020	0.019	0.014	0.010	0.009	0.001
Michigan melon	Cucurbits, soil chemigation/drench	0.018	0.017	0.016	0.012	0.007	0.005	0.002
New Jersey melon	Cucurbits, soil chemigation/drench	0.059	0.056	0.054	0.040	0.023	0.017	0.004
Idaho potato	Potato, in-furrow	0.03	NA[b]	0.025	0.019	0.012	0.008	0.002
Maine potato	Potato, in-furrow	0.21	NA[b]	0.20	0.16	0.12	0.09	0.03

a. For application rates (single and annual), application interval, and number of applications considered for each use pattern, see Supporting Information Section S7 (Table S7-3).
b. NA - 48 h average EECs are not a standard calculation in EPA's Tier II modeling.

Table 6 Summary of refined 90th percentile exposure predictions for golf course turf use (µg a.i./L) with and without percent treated area (PTA) refinement

Golf Course Scenario[a]	***Use Pattern***	***Peak***	***96-hr Average***	***21-day Average***	***60-day Average***	***90-day Average***	***Annual Average***
Florida Turf	Ground app., 100% PTA	0.38	0.34	0.24	0.13	0.09	0.02
	Ground app., 12.2% PTA	0.046	0.042	0.029	0.016	0.011	0.003
Pennsylvania Turf	Ground app., 100% PTA	0.66	0.61	0.46	0.31	0.25	0.07
	Ground app., 10.1% PTA	0.067	0.061	0.046	0.031	0.025	0.007

Notes:
a.

[a]For application rates (single and annual), application interval, and number of applications considered for each use pattern, see Supporting Information Section S9 (Table S9-1).

Table 7 Summary of refined 90th percentile exposure predictions for residential use patterns (California Residential/California Impervious Scenario) (µg a.i./L) with and without percent treated area (PTA) refinement

Use Pattern[a]	*Peak*	*96-hr Average*	*21-day Average*	*60-day Average*	*90-day Average*	*Annual Average*
Turf, 2 Apps., 100% PTA	0.60	0.55	0.41	0.25	0.19	0.05
Turf, 2 Apps, 2.08% PTA	0.012	0.011	0.008	0.005	0.004	0.001
Turf, 1 App, 100% PTA	0.53	0.48	0.36	0.22	0.17	0.04
Turf, 1 App., 2.08% PTA	0.011	0.010	0.007	0.005	0.003	0.001
Ornamentals., 100% PTA	0.33	0.30	0.23	0.14	0.11	0.03
Ornamentals, 2.08% PTA	0.007	0.006	0.005	0.003	0.002	0.001
Perimeter, 100% PTA	0.36	0.33	0.25	0.16	0.11	0.03
Perimeter, 2.08% PTA	0.008	0.007	0.005	0.003	0.002	0.001

[a]For application rates (single and annual), application interval, and number of applications considered for each use pattern, see Supporting Information Section S9 (Table S9-1).

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Table 8 Summary of refined 90th percentile exposure predictions for nursery use patterns (µg a.i./L) with and without percent treated area (PTA) refinement

Scenario[a]	*Peak*	*96-hr Average*	*21-day Average*	*60-day Average*	*90-day Average*	*Annual Average*
California Nursery, 100% PTA	0.22	0.21	0.14	0.08	0.06	0.02
California Nursery, 1.16% PTA	0.0026	0.0024	0.0017	0.0010	0.0007	0.0002
Florida Nursery, 100% PTA	4.42	4.05	2.97	1.68	1.24	0.29
Florida Nursery, 0.41% PTA	0.018	0.017	0.012	0.007	0.005	0.001
Michigan Nursery, 100% PTA	1.76	1.63	1.33	0.95	0.79	0.27
Michigan Nursery, 1.7% PTA	0.030	0.028	0.023	0.016	0.013	0.005
New Jersey Nursery, 100% PTA	2.58	2.38	1.82	1.16	0.88	0.23
New Jersey Nursery, 1.24% PTA	0.044	0.040	0.031	0.020	0.015	0.004
Oregon Nursery, 100% PTA	0.92	0.86	0.73	0.53	0.41	0.11
Oregon Nursery, 0.54% PTA	0.016	0.015	0.012	0.009	0.007	0.002
Tennessee Nursery, 100% PTA	2.69	2.47	1.85	1.11	0.89	0.21
Tennessee Nursery, 0.68% PTA	0.046	0.042	0.031	0.019	0.015	0.003

a. For application rates (single and annual), application interval, and number of applications considered for each use pattern, see Supporting Information Section S9 (Table S9-1).

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Table 9 Acute risk categories for agricultural use patterns

Scenario	***AUC (%)***	***Risk Category Based on AUC***
California citrus	0.143	*De minimis*
California lettuce	0.083	*De minimis*
California tomato	0.164	*De minimis*
Florida cabbage	0.0116	*De minimis*
Florida citrus	0.189	*De minimis*
Florida cucumber	0.0331	*De minimis*
Florida peppers	0.0373	*De minimis*
Florida tomato	0.0231	*De minimis*
Idaho potato	0.0253	*De minimis*
Maine potato	0.202	*De minimis*
Michigan melon	0.0229	*De minimis*
Mississippi soybean	0.342	*De minimis*
New Jersey melon	0.0638	*De minimis*
Pennsylvania tomato	0.0113	*De minimis*

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Table 10 Acute risk categories for non-agricultural use patterns with and without percent treated area (PTA) refinement

Scenario	***Percent Treated Area (PTA) Assumed***	***AUC (%)***	***Risk Category***
Golf Course (Florida Turf)	100% PTA	0.48	*De minimis*
	Refined PTA	0.0609	*De minimis*
Golf Course (Pennsylvania Turf)	100% PTA	0.855	*De minimis*
	Refined PTA	0.086	*De minimis*
Residential Turf, 2 Apps	100% PTA	0.745	*De minimis*
	Refined PTA	0.0271	*De minimis*
Residential Turf, 1 App	100% PTA	0.646	*De minimis*
	Refined PTA	0.0229	*De minimis*
Residential Ornamentals	100% PTA	0.413	*De minimis*
	Refined PTA	0.0162	*De minimis*
Residential Perimeter	100% PTA	0.451	*De minimis*
	Refined PTA	0.0182	*De minimis*
Nursery (California)	100% PTA	0.629	*De minimis*
	Refined PTA	0.007	*De minimis*
Nursery (Florida)	100% PTA	5.09	Low
	Refined PTA	0.0297	*De minimis*
Nursery (Michigan)	100% PTA	2.66	Low
	Refined PTA	0.0577	*De minimis*
Nursery (New Jersey)	100% PTA	3.34	Low
	Refined PTA	0.0617	*De minimis*
Nursery (Oregon)	100% PTA	1.38	*De minimis*
	Refined PTA	0.0326	*De minimis*
Nursery (Tennessee)	100% PTA	3.37	Low
	Refined PTA	0.062	*De minimis*

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Table 11 Chronic risk categories for agricultural use patterns generated with the chronic lab-based SSD (Chronic I) and the chronic higher tier TSD (Chronic II)

	Chronic I:		***Chronic II:***	
Scenario	***AUC (%)***	***Risk Category Based on AUC***	***AUC (%)***	***Risk Category Based on AUC***
California citrus	3.41	Low	0.0649	*De minimis*
California lettuce	2.54	Low	0.000628	*De minimis*
California tomato	3.25	Low	0.0632	*De minimis*
Florida cabbage	0.383	*De minimis*	0.000110	*De minimis*
Florida citrus	3.82	Low	0.0734	*De minimis*
Florida cucumber	0.897	*De minimis*	0.00306	*De minimis*
Florida peppers	1	*De minimis*	0.00238	*De minimis*
Florida tomato	0.633	*De minimis*	0.00128	*De minimis*
Idaho potato	1.13	*De minimis*	5.00E-08	*De minimis*
Maine potato	6.59	Low	5.00E-08	*De minimis*
Michigan melon	0.696	*De minimis*	2.93E-04	*De minimis*
Mississippi soybean	7.83	Low	0.031	*De minimis*
New Jersey melon	1.76	Low	0.00118	*De minimis*
Pennsylvania tomato	0.361	*De minimis*	8.81E-05	*De minimis*

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Table 12 Chronic risk categories for non-agricultural use patterns generated with the chronic lab-based SSD (Chronic I) and the chronic higher tier TSD (Chronic II) with and without percent treated area (PTA) refinement

Scenario	***Percent Treated Area (PTA) Assumed***	***Chronic I:***		***Chronic II:***	
		AUC (%)	***Risk Category Based on AUC***	***AUC (%)***	***Risk Category Based on AUC***
Golf Course (Florida Turf)	100% PTA	8.3	Low	0.317	*De minimis*
	Refined PTA	1.79	Low	0.0016	*De minimis*
Golf Course (Pennsylvania Turf)	100% PTA	13.5	Intermediate	0.721	*De minimis*
	Refined PTA	2.66	Low	0.00154	*De minimis*
Residential Turf, 2 Apps	100% PTA	14.6	Intermediate	0.0994	*De minimis*
	Refined PTA	0.758	*De minimis*	5E-08	*De minimis*
Residential Turf, 1 App	100% PTA	13	Intermediate	0.0868	*De minimis*
	Refined PTA	0.669	*De minimis*	5E-08	*De minimis*
Residential Ornamentals	100% PTA	10	Intermediate	0.00692	*De minimis*
	Refined PTA	0.482	*De minimis*	5E-08	*De minimis*
Residential Perimeter	100% PTA	10.8	Intermediate	0.00536	*De minimis*
	Refined PTA	0.527	*De minimis*	5E-08	*De minimis*
Nursery (California)	100% PTA	14	Intermediate	0.118	*De minimis*
	Refined PTA	0.388	*De minimis*	5E-08	*De minimis*
Nursery (Florida)	100% PTA	38.5	High	12.6	Intermediate
	Refined PTA	0.906	*De minimis*	5E-08	*De minimis*
Nursery (Michigan)	100% PTA	33.3	High	3.91	Low
	Refined PTA	1.92	Low	5E-08	*De minimis*
Nursery (New Jersey)	100% PTA	33.9	High	6.85	Low
	Refined PTA	2.13	Low	5.25E-08	*De minimis*
Nursery (Oregon)	100% PTA	23.1	Intermediate	0.978	*De minimis*
	Refined PTA	1.14	*De minimis*	5E-08	*De minimis*
Nursery (Tennessee)	100% PTA	32.8	Intermediate	6.72	Low
	Refined PTA	2.07	Low	5.37E-08	*De minimis*

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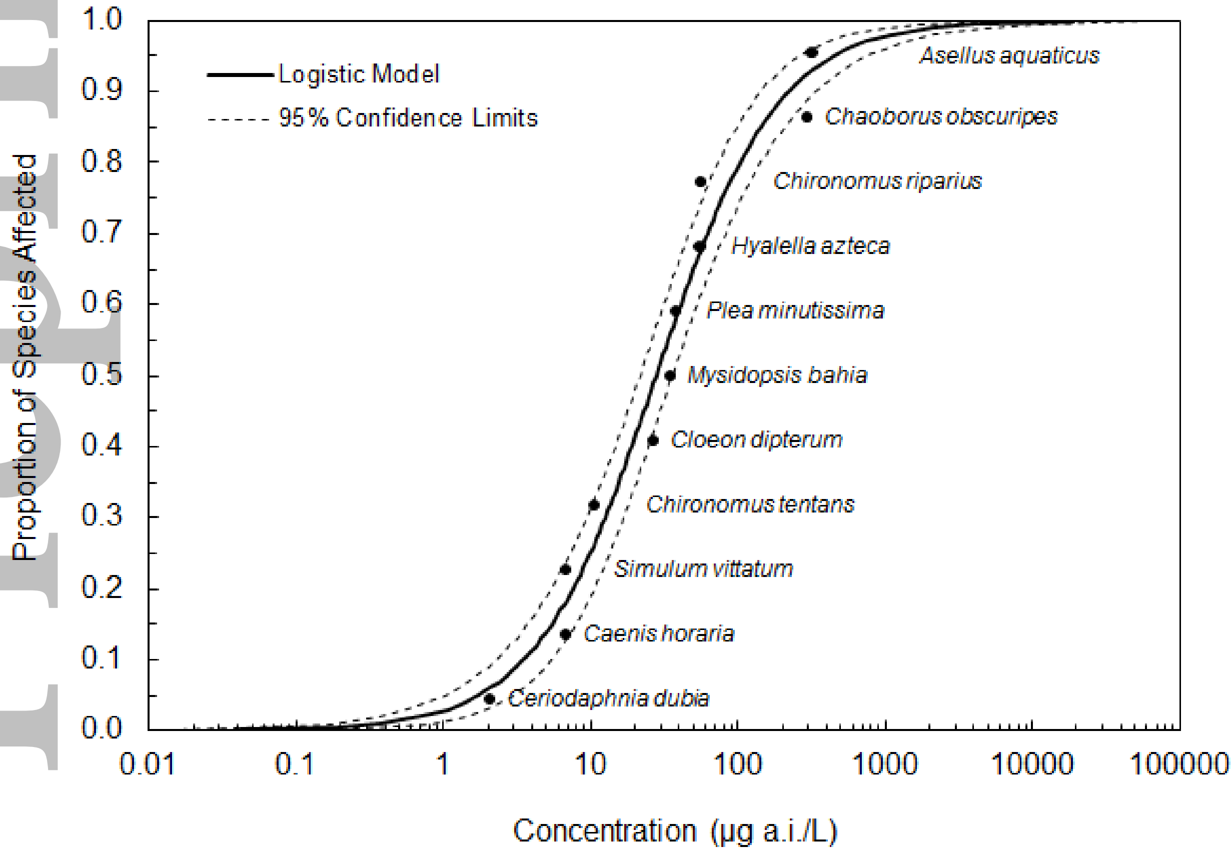

Figure 1

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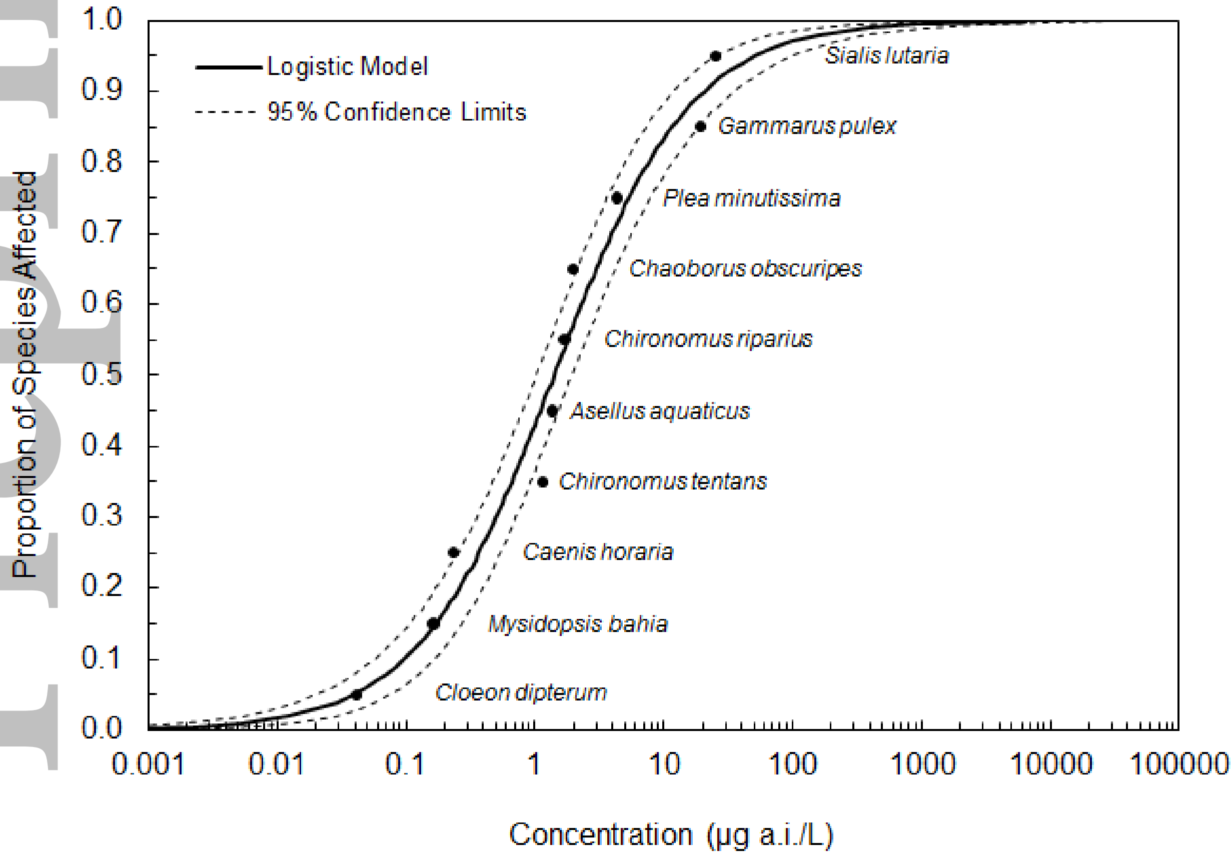

Figure 2

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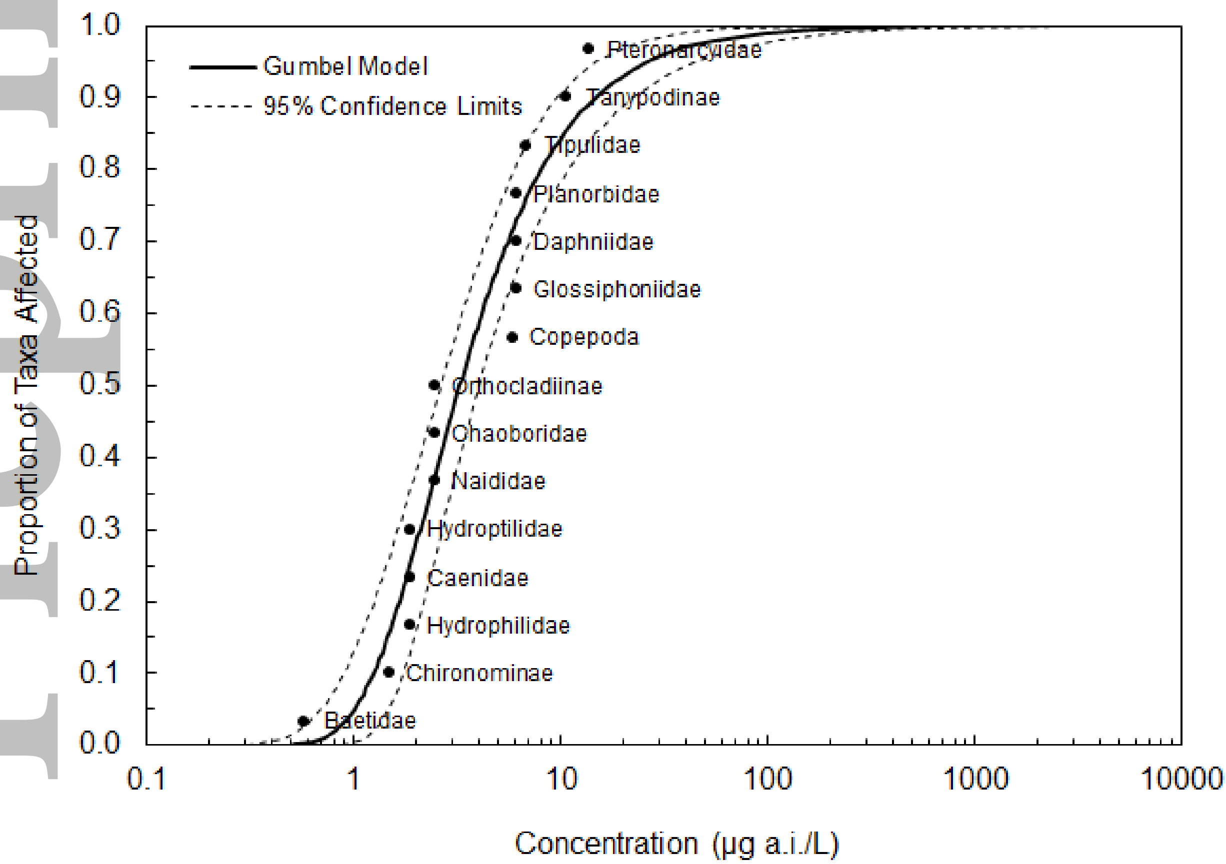

Figure 3

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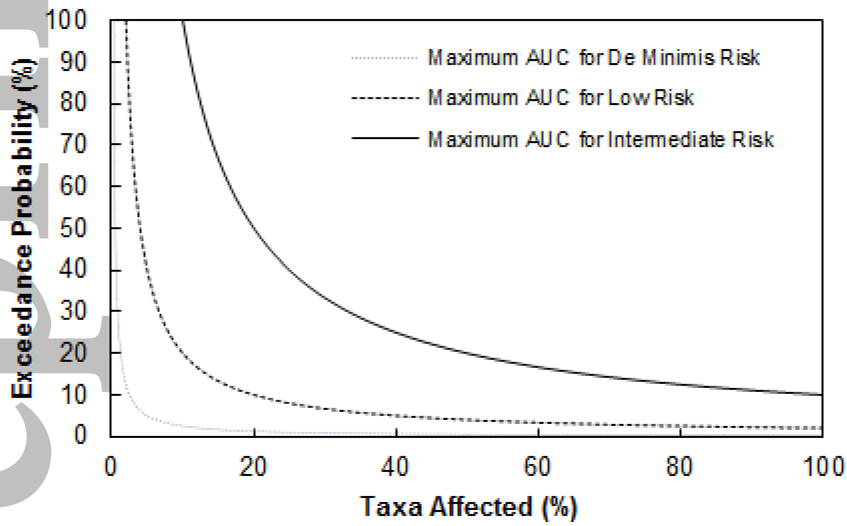

Figure 4

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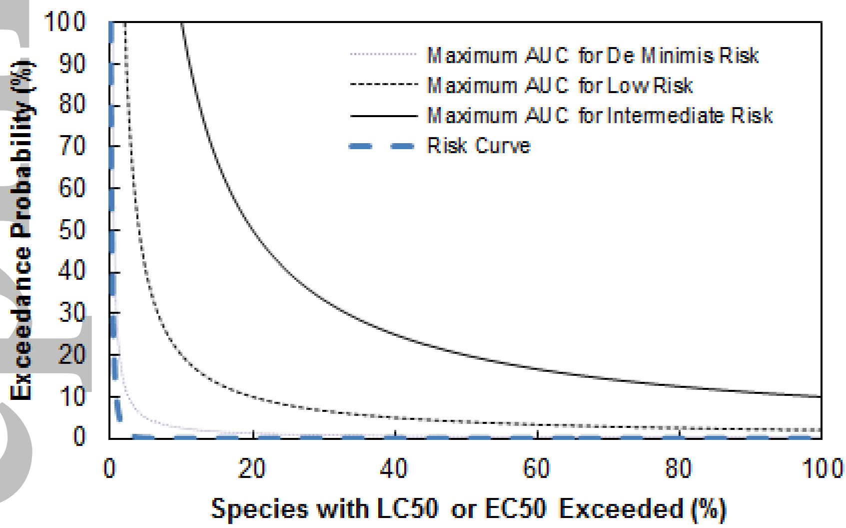

Species with LC50 or EC50 Exceeded (%)	Exceedance Probability (%)
5	0.11
10	<0.00333
25	<0.00333
50	<0.00333
75	<0.00333
90	<0.00333
95	<0.00333

Risk Summary	
AUC (%)	0.342
Risk Category	*De minimis*

Figure 5

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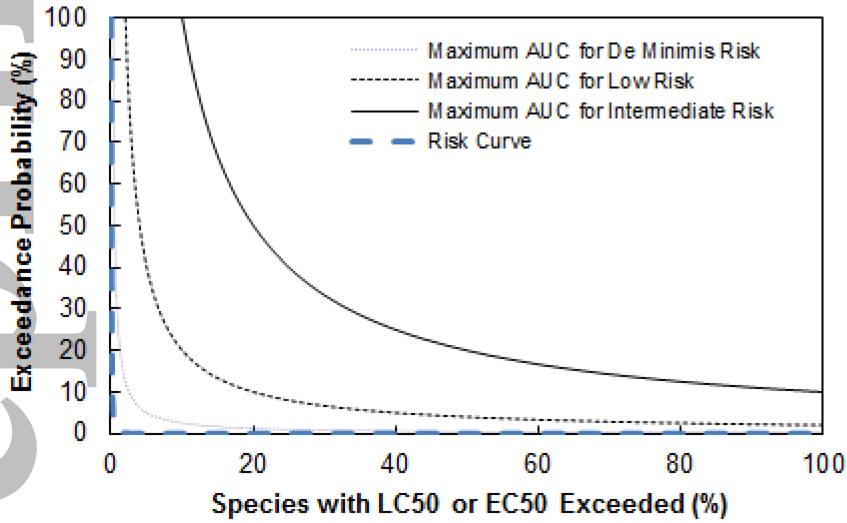

Species with LC50 or EC50 Exceeded (%)	**Exceedance Probability (%)**
5	0.00333
10	<0.0000333
25	<0.0000333
50	<0.0000333
75	<0.0000333
90	<0.0000333
95	<0.0000333

Risk Summary	
AUC (%)	0.0860
Risk Category	*De minimis*

Figure 6

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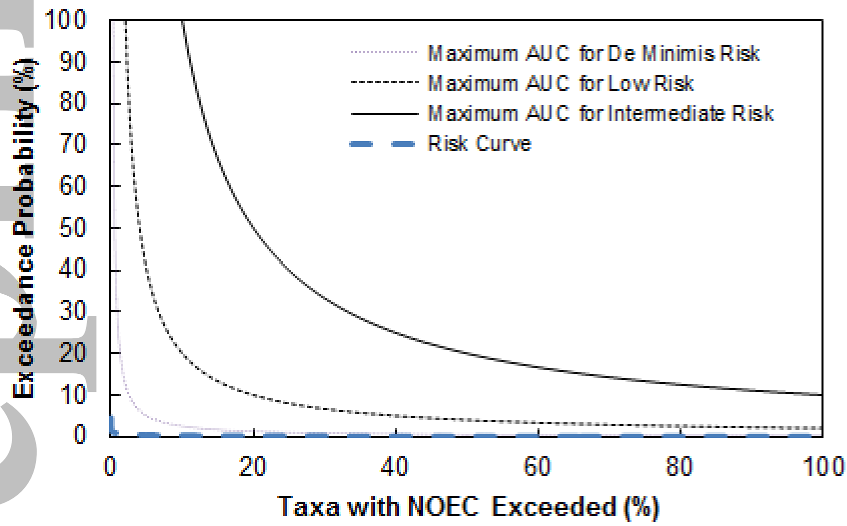

Taxa with NOEC Exceeded (%)	Exceedance Probability (%)
5	0.327
10	0.210
25	0.0767
50	0.0133
75	0.00333
90	<0.00333
95	<0.00333

Risk Summary	
AUC (%)	0.0734
Risk Category	*De minimis*

Figure 7

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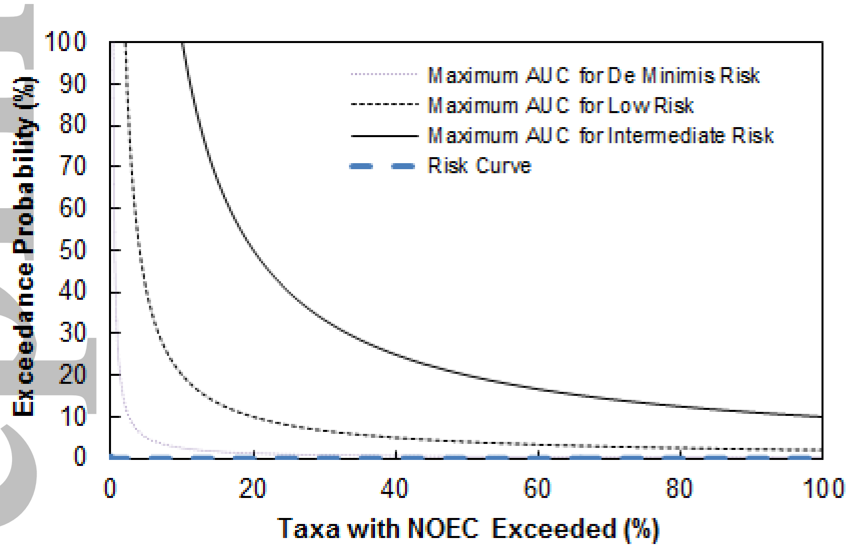

Taxa with NOEC Exceeded (%)	Exceedance Probability (%)
5	0.00667
10	0.00333
25	<0.0000333
50	<0.0000333
75	<0.0000333
90	<0.0000333
95	<0.0000333

Risk Summary	
AUC (%)	0.00160
Risk Category	*De minimis*

Figure 8

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