Chapter 3 – Imidacloprid Exposure Characterization

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Environmental Transport and Fate Characterization

Imidacloprid has a high solubility, low octanol-water partitioning coefficient, low vapor pressure, and low Henry’s Constant (**Table 3-1**). These data suggest that imidacloprid has a low potential for volatilization and bioaccumulation. However, the chemical will be readily soluble and thus available for leaching and movement with run-off water. The chemical will initially enter the environment via direct application (*e.g.*, as liquid sprays, dusts, seed coatings, granular formulations) to use sites (*e.g.*, seed treatment, soil, foliage). It may move off-site via spray drift, dissolved in runoff, and/or as residue sorbed to eroded sediment.

Table 3-. Physical and Chemical Properties of Imidacloprid

|  |  |
| --- | --- |
| ***Property*** | ***Value*** |
| Chemical Structure/Name | Diagram, schematic  Description automatically generated  1-(6-chloro-3-pyridin-3-ylmethyl)-N-nitroimidazolidin-2- ylidenamine  **Smiles code:** C1CN(/C(=N\[N+](=O)[O-])/N1)CC2=CN=C(C=C2)Cl |
| CAS Number | 138261-41-3 |
| Molecular Formula | C9H10ClN5O2 |
| Molecular Weight (CAS No.) | **255.7** g/mole (13826-41-3) |
| Water Solubility @ 20 oC | **580-610** mg/L (ppm) |
| Octanol: Water Coefficient Kow | **3.7 @ 21** oC |
| Vapor pressure (Henry’s Law Constant) | **1.5 x 10-9** torr (**9.9 x 10-13** atm m3 mol-1) @ 20 oC |

Imidacloprid is hydrolytically stable. The chemical is highly susceptible to photodegradation in water with an observed half-life of 0.2 days. Aerobic and anaerobic aquatic transformation are expected to contribute to dissipation of imidacloprid reaching aquatic systems by run-off and drift. Metabolism appears to be more pronounced in anaerobic conditions (t½ = 33 days) compared to aerobic conditions (t½ >30 to 159 days). In contrast, imidacloprid reaching the soil system by direct application and wash-off is expected to be highly persistent (t½ 132 to 608 days). Persistence in soils may lead to accumulation over the years with repeated applications. However, the magnitude of soil accumulation is expected to be highly affected by other important routes of dissipation including leaching, run-off and plant up-take which are expected to reduce this accumulation.

A summary of available environmental fate data for imidacloprid is provided in **Table 3-2**.

Table 3-. Environmental Fate Data for Imidacloprid

| **Study/Property** | **Value(s)1** | **Major degradate(s)** | **Comments** | **MRID2** |
| --- | --- | --- | --- | --- |
| Hydrolysis t½ | **Stable** @ pH 5, 7 and hydrolyzed slowly (Extrapolated t½= **355 d**) in sterile alkaline solutions @ pH 9 | None |  | 420553-37 (A) |
| Aqueous Photolysis t½ | **0.2 days** | Guanidine Max 17% and urea compound Max 10% @ End of study= EOS | 1. UV spectrum of the chemical has a maximum absorption at 269 nm, therefore degradation by sunlight is expected  2. Under natural sunlight, in a dilute aqueous solution in the greenhouse: 60% of the chemical degraded within 4 hours supporting the results of the study | 422563-76 (A) |
| Soil Photolysis t½ | **171 days** in a sandy loam soil from Kansas (pH= 5.2; O.C= 1.4% and CEC= 22 meq/100 g) | None |  | 422563-77 (A) |
| Aerobic soil t½ @ 20 oC | **608 days** (SFO\*) in a sandy loam soil from Kansas (pH= 4.8; O.C= 1.4% and CEC= 16 meq/100 g) | None | Extrapolated value because parent reached only 62% @ EOS  *Mineralization to CO2:* Max 7.4% @ EOS (366 days) | 420735-01 (A) |
| Aerobic soil t½ @ 20 oC | **172 days** (Slow DFOP) in BBA 2.2, a loamy sand soil from Germany (pH= 5.5; O.C= 2.2% and CEC= 10 meq/100 g) | None | Extrapolated value because parent reached only 63% @ EOS)  *Mineralization to CO2:* Max 10% @ EOS (100 days) | 452393-01 (A) |
| Aerobic soil t½ @ 20 oC | **193 days** (SFO) in Hoefchen, a silt soil from Germany (pH= 5.3; O.C= 1.2% and CEC= 11 meq/100 g) | None | Extrapolated value because parent reached only 67% @ EOS)  *Mineralization to CO2:* Max 6.4% @ EOS (100 days) | 452393-02 (A) |
| Aerobic soil t½ @ 22 oC | **336 days** (SFO) in Monheim 1, a sandy loam soil from Germany (pH= 5.3; O.C= 1.2% and CEC= 11 meq/100 g) | None | Extrapolated value because Parent reached 52% @ EOS  *Mineralization to CO2:* Max 6.4% @ EOS (366 days) | 452393-03 (A) |
| Aerobic soil t½ @ 20 oC | **139 days** (slow DFOP\*) in Sarotti soil (silt loam, pH 7.0, O.C= 1.46% and CEC= 13 meq/100 g) soil from Germany  **242 days** (slow DFOP\*) in Laacherhof (sandy loam, pH 6.2, O.C= 1.88% and CEC= 9 meq/100 g) soil from Germany  **332 days** (slow DFOP) in Wurmwiese soil (sandy loam, pH 5.4, O.C= 1.69% and CEC= 10 meq/100 g) soil from Germany  **177 days** (slow DFOP\*) in Hoefchen am Hohenseh 4a soil (silt loam, pH 6.5, O.C= 2.59% and CEC= 14 meq/100 g) soil from Germany | None | Extrapolated value because Parent reached 40% in the 1st soil, 50% in the 2nd soil, 57% in the 3rd soil and 38% in the 4th soil @ EOS  *Mineralization to CO2:* Max 12-28% @ EOS (120 days) | 498358-02, |
| Anaerobic Aquatic t½ @ 22 oC | **33 days** (SFO) in pond water sediment system from Stanly, Kansas (Water: Total organic carbon (TOC) 5 mg/L; Sediment: silt loam, pH 6.9, O.C= 3.15% and CEC= 14 meq/100 g) | Guanidine= Max 21% @ 60 d declined to 16% @ EOS | *Mineralization to CO2:* Max 0.2-0.5% @ 249 d to EOS (358 days) | 422563-78 (S) |
| Aerobic Aquatic t½ @ 22 oC | **32 days** (SFO) in an orchard ditch water: loamy silt sediment system (Water: pH 8.4 and TOC= 5 mg/L; Sediment: OC%= 4.1%) from IJzendoorn, Netherlands  **159 days** (SFO) in a re-cultivated quarry water: loamy sand sediment system (water: pH 8.1 and TOC= 4 mg/L; sediment: OC%= 0.9%) from Lienden, Netherlands | Guanidine= 9-12% @ EOS | *Mineralization to CO2:* Max 1.4-2% @ EOS (92 days) | 484169-01 (S) |
| Aerobic Aquatic t½ @ 22 oC | **>30 days** () in a pond water: silty clay sediment system (water: pH 8.5 and TOC= 4 mg/L; sediment: pH 7.6 and OC%= 2.1%) from Kansas, USA | None | Parent was 84% @ EOS  *Mineralization to CO2:* Max 0.7% @ EOS (30 days) | 484169-02 (S) |
| Koc (L Kg -1) | **Parent: Average= 266** (n=15) ranging from 98-487 in soils with varied texture, Clay= 1 to 43%, Organic carbon (O.C) = 0.23 to 3.95%, pH= 4.5 to 7.8, and Cation exchange capacity (C.E.C) = 4 to 41 meq/100 g  **Guanidine Metabolite: Average= 742** (n=4) ranging from 327 to 942 in soils with varied texture (Sand, Loamy sand, Sandy loam and Loam, O.C= 0.23 to 1.51%, pH= 5.1 to 6.5, and C.E.C= 4 to 16 meq/100 g |  |  | 425208-2(A) |
| 1 Values for half-lives were estimated as per NAFTA degradation kinetic: SFO model= Single Order; DFOP model= Double First Order; and IORE model= Indeterminate Order Rate Equation.  2 Study classification: A= Acceptable; S= Supplemental | | | | |

# Identification of Transformation Products of Concern

Imidacloprid is expected to resist degradation in the soil system producing only minor amounts of degradation products (maximum of 2%). Formed degradates include olefin, 5-keto-urea isomers, nitrosamine, and guanidine. Formed degradates preserve the structure of their parent. Eventually, degradation products degrade into the 6-chloronicotinic acid and varied amounts CO2 (5-28%) and bound residues. In aquatic systems, imidacloprid is expected to be affected by biotic metabolism which is more pronounced under anaerobic conditions than aerobic conditions. Guanidine is the major degradation product as it reached a maximum of 9% of the applied under aerobic conditions and 21% of the applied under anaerobic conditions. Under aerobic aquatic conditions, three minor degradates appear to form namely, 5-hydroxy (Max= 3%), the transitional degradate DIJ 9646-2 and the terminal degradate 6- chloronicotinic acid (Max= 4%). Mineralization to CO2 ranged from a maximum of 0.5% under anaerobic conditions to a maximum of 2% under aerobic conditions. Chemical structures and names of the major degradates of imidacloprid (Maximum formation ≥10%) are presented in Table 3-3.

Table 3-3. Major Degradation Products of Imidacloprid

|  |  |  |  |
| --- | --- | --- | --- |
|  | **Maximum Formation (% of Applied Radioactivity)** | | |
| **Degradate Name and Structure** | **Aqueous Photolysis (MRID 422563-76)** | **Aerobic Aquatic**  **(MRID 484169-01)** | **Anaerobic Aquatic (MRID 422563-78)** |
| Guanidine | 17% | 9-12% | 21% |
| Imidacloprid Urea | 10% | Not Identified | Not Identified |

As discussed above, imidacloprid may degrade into various degradation products through multiple pathways. Metabolites identified from aerobic soil metabolism studies include IMI-olefin, nitrosamine, guanidine, and 5-keto urea isomers. The formation rates of the degradates from this pathway do not exceed 2% of the applied and are therefore considered to be minor. Conversely, in the aqueous photolysis pathway, imidacloprid degrades to the guanidine and urea metabolites at rates up to 17 and 10% of the applied residues, respectively. Formation rates for guanidine also reach 12 and 21% for the aerobic and anaerobic aquatic metabolism pathways, respectively. Available acute toxicity data for the guanidine and urea degradates indicate their toxicity is at least 3 orders of magnitude less than parent imidacloprid (**Chapter 2**). Therefore, the stressor of concern for this assessment is determined to be parent imidacloprid alone.

# Measures of Aquatic Exposure

Maximum application rates/number of applications and minimum application retreatment intervals are modeled using PWC or PFAM to estimate the exposure to imidacloprid based on current labeled use (**APPENDIX 1-2**). To streamline the assessment, some use patterns were grouped based on similar application parameters, use sites and relevant aquatic modeling scenario.

Imidacloprid-specific modeling scenarios are used for modeling each use. This includes the selection of scenarios and agronomic practices (*e.g.*, applications methods, dates). **APPENDIX 3-1** includes model use input parameters as well as the justification for selecting these parameters. The general approaches used in determining potential exposure are described below.

## Aquatic Exposure Models

Aquatic exposures (surface water and benthic sediment pore water) are quantitatively estimated for representative imidacloprid uses included on the labels (**APPENDIX 1-2**) by HUC 2 Regions (**Figure 3-1**) and by aquatic bins (2-7).

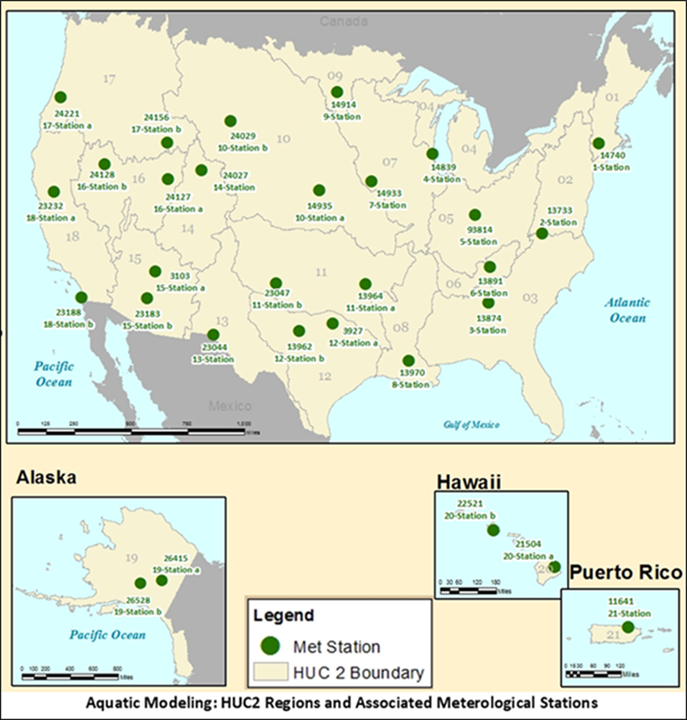


Figure 3-1. Hydrologic Unit Code (HUC) 2-Digit Regions and Associated Metrological Data

Several models are available to use to estimate pesticide concentrations in surface water. The primary model used in this assessment is the Pesticide Root Zone Model (PRZM5) and the Variable Volume Water Model (VVWM)[[1]](#footnote-2) in the Pesticides in Water Calculator (PWC). As mentioned elsewhere, the flowing aquatic bins include bin 2 low flow, bin 3 moderate flow, and bin 4 high flow. The static aquatic bins include bin 5 low volume, bin 6 moderate volume, and bin 7 high volume. Additional information on aquatic bins is available in **ATTACHMENT 3-1**.

Aquatic bin 1 represents aquatic habitats associated with terrestrial habitats (*e.g.*, riparian zones, seasonal wetlands) and is simulated using the PRZM5/VVWM and the Plant Assessment Tool (**PAT,** **Section 3.5**). Aquatic bins 8 and 9 are intertidal and subtidal near shore habitats, respectively, and aquatic bin 10 is the offshore marine habitat. EFED does not currently have standard conceptual models designed to estimate EECs for these estuarine/marine systems. EFED and the Services have assigned surrogate freshwater flowing or static systems to evaluate exposure for these estuary and marine bins. Aquatic bin 5 will be used as surrogate for pesticide exposure to species in tidal pools; aquatic bins 2 and 3 will be used for exposure to species at low and high tide, and aquatic bins 4 and 7 will be used to assess exposure to marine species that occasionally inhabit offshore areas.

EFED modeled these flowing and static bins in previous Biological Evaluations (BEs) by using watershed drainage areas developed from flowing waterbody relationships and static bin runoff estimates (USEPA, 2016a; USEPA, 2016b; USEPA, 2016c). However, the results have not been as expected. In short, one would expect daily concentrations in flowing bins to be lower than those in static bins, given that water, and any pesticide, is flowing out of the system. However, in most of the modeling runs, the EECs for the flowing bins were higher, and in many cases an order of magnitude higher, than the static bins. In large part, the higher EECs were the result of modeling a large watershed using assumptions that were designed to simulate a smaller watershed. For example, it was assumed that the watershed was entirely treated on the same day. For a large watershed, it is not expected that this assumption is appropriate. For the static waterbodies, the watershed-to-waterbody ratio was not really known but was derived by determining the watershed size needed to generate runoff sufficient to fill the waterbody. This relationship relied on modeling conducted using topography, surface soil characteristics, and precipitation specific to the PWC model and scenarios considered highly vulnerable to surface water runoff. As a result, there was much uncertainty with the EECs that were being generated from the modeling in the previously completed BEs.

For imidacloprid, when using PWC, EFED has relied on two standard waterbodies which have been traditionally used in EFED to estimate EECs for the various bins. The standard farm pond was used to develop EECs for the medium and large static bins (*e.g.*, bins 6 and 7) and the index reservoir for the medium and large flowing bins (*e.g.*, bins 3 and 4). For the smallest flowing and static bins (bin 2 and 5), EFED derived edge of field estimates from the PRZM daily runoff file (*e.g.*, ZTS file). **Table 3-4** provides a crosswalk of the bins and how they were modeled.

Table 3-. Aquatic Bin, Modeled Waterbody Crosswalk

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Aquatic Bin** | **Description** | **Width (m)** | **Length (m)** | **Depth (m)** | **Flow (m3/s)** | **Waterbody Used for Modeling** |
| 1 | Wetland | 64 | 157 | 0.15 | Variable1 | Custom |
| 2 | Low-flowing waterbody | 2 | Field2 | 0.1 | 0.001 | Edge-of-field |
| 3 | Medium-flowing waterbody | 8 | Field2 | 1 | 1 | Index reservoir |
| 4 | High-flowing waterbody | 40 | Field2 | 2 | 100 | Index reservoir |
| 5 | Low-volume, static waterbody | 1 | 1 | 0.1 | N/A | Edge-of-field |
| 6 | Medium-volume, static waterbody | 10 | 10 | 1 | N/A | Farm pond |
| 7 | High-volume, static waterbody | 100 | 100 | 2 | N/A | Farm pond |

1 The depth and flowrate in this waterbody is variable, depending on rainfall.

2 The habitat being evaluated is the reach or segment that abuts or is immediately adjacent to the treated field. This habitat is assumed to run the entire length of the treated area. NA – not applicable.

While the standard farm pond is bigger than bin 6, the EECs estimated for bin 6 in previous BEs were close to those generated for bin 7, and so an economy of modeling was deemed appropriate.

While the index reservoir has a much lower effluent flowrate than bins 3 and 4, it has been used as a vetted flow-through waterbody for EFED for years, with an accepted watershed-to-waterbody ratio developed for an actual vulnerable watershed (Shipman Reservoir, Shipman, IL) and has been reviewed by a previous Federal Insecticide Fungicide Rodenticide Act (FIFRA) Scientific Advisory Panel (SAP) (Jones *et al*, 1998). EFED expects the EECs that are generated using the index reservoir to be a conservative surrogate for those observed in bins 3 and 4. The watershed area associated with the index reservoir is roughly an order of magnitude smaller than the average area for a HUC 12 (the smallest areal delineation for an aquatic species range), but within the range of minimum and maximum values (9.54x107 m2, 2.08x103 – 9.24x109 m2).

Lastly, bins 2 and 5 are very small waterbodies and the EECs in them would be reflective of concentrations in a headwater stream or a standing puddle that received runoff at the edge of a treated field. As such, edge-of-field concentrations were estimated and used as a surrogate for EECs in these waterbodies.

More detailed information can be found in **ATTACHMENT 3-1** (Background Document: Aquatic Exposure Estimation for Endangered Species).

## HUC and Use Site Crosswalk

The National Agricultural Statistics Census of Agriculture 2012 (NASS) data along with the Cropland Data Layer (CDL) were used to determine which crops would be modeled within each represented HUC 2. Additionally, specific geographic limitations on how a product may be applied to particular crops were considered when determining what rates would be simulated for different HUC 2 regions. For example, spray drift buffers are required, and spray drift fractions were adjusted using AgDRIFT model.

If the NASS data indicated any acreage of a crop was grown in a specific HUC 2, it was assumed that the crop was grown in that HUC 2, and aquatic EECs were generated for these HUC 2 regions for that crop. If there were no reported NASS cropped acres grown within a particular HUC 2, aquatic EECs for that HUC 2 region and use pattern were not determined. A crop use layer-HUC 2 region matrix for imidacloprid is provided in **APPENDIX 3-1**. Limited NASS data are available for Alaska, Hawaii, and Puerto Rico, and some assumptions on which crops would be simulated in those HUC 2 regions were made.

## Scenario Selection

A PWC scenario was developed for each landcover class and HUC 2 where crops in that landcover were grown based on the NASS 2012 census data. **APPENDIX 3-1** provides a crosswalk between the use site and the landcover used to represent the use site as well as which HUC 2 regions were evaluated for each use pattern. An explanation of how the PWC scenario matrix was developed is provided in **ATTACHMENT 3-1**.

## Application Practices

### Application Method

During application of pesticides, methods of application as well as product formulation used by an applicator can impact the magnitude of off-site transport of the active ingredient. Label directions (such as spray drift buffers and droplet size restrictions, application equipment, and agronomic practices such as soil incorporation) as well as product formulation are considered as part of the development of the use scenario modeled.

There are several different types of imidacloprid applications used in both agricultural and non-agricultural settings. Applications occur from aircraft and varied ground sprayers including airblast sprayers. Imidacloprid is formulated as granules; ready-to- use liquids; emulsifiable concentrates; flowable concentrates; water soluble packaging; pelleted/tableted products; water dispersible granules; wettable powders; impregnated materials; dust; and pressurized liquids. End use products are applied as: liquid spray or drench; broadcast or in-band granules; broadcast or in station baits; and as seed coating.

Imidacloprid applications may occur at different times throughout the year including multiple applications to the same crop. When multiple types of applications are allowed on a crop within one calendar year, such as pre-plant along with those occurring throughout the growing season, all applications are simulated considering the appropriate application timing specified by label directions and restrictions.

### Spray Drift

Imidacloprid labels includes spray buffers from lakes; reservoirs; rivers; permanent streams, marshes or natural ponds; estuaries and commercial fish farm ponds. A 25 feet buffer is required for ground applications and 150 feet buffer for aerial applications. Existence of label buffers requires calculations of the reduced drift fractions that will be used for modeling. AgDRIFT[[2]](#footnote-3) was used for these calculations and the results are summarized in **Table 3-5** for liquid spray applications noting that spray drift for dry applications is assumed to be zero.

Table 3-. Estimated Spray Drift Fractions for Different Aquatic Bins and Application Methods

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Bin** | | | | | **Spray Drift Fraction (unitless)** | | |
| **Aquatic Bin** | **PWC Bin Number** | **Generic Habitat** | **Depth (m)1** | **Width (m)1** | **Aerial** | **Airblast** | **Ground** |
| 1 | 10 | Wetland | 0.15 | 64 | 0.0385 | 0.015 | 0.0267 |
| 4 | 4 | Reservoir | 2.74 | 82 | 0.0421 | 0.021 | 0.0322 |
| 7 | 7 | Pond | 2 | 64 | 0.0385 | 0.015 | 0.0267 |

1parameters correspond to the input values used in PWC modeling.

Edge of field (EOF) concentrations from bin 4 are used as a surrogate for aquatic bins 2 and 5.

Aquatic bin 4 is used as a surrogate for aquatic bin 3.

### Application Timing

In order to achieve optimum pest control, directed or broadcast foliar sprays of imidacloprid are recommended to be at the earliest threshold for target pests. Optimum pest control for soil applications is expected to result from application to the root zone of the plants at an early time so that is available for earlier protection to the developing plant. In determining application timing for soil and foliar applications, many factors were considered in selecting the application windows for imidacloprid use patterns. The factors were derived from label information on timing/length of application including: crop stage(s) at time of application; pre-harvest intervals; restrictions related pollinator protection; and the time needed to complete applications (if more than one application is permitted). Other sources of information on crop stages included crop profiles (https://ipmdata.ipmcenters.org/#cropprofiles), agricultural extension bulletins and the literature. For details on application date selection for use of imidacloprid, see **APPENDIX 1-3** and **APPENDIX 3-1**.

## Special Agricultural Considerations

### Multiple Crop-cycles Per Year

Some labels permit applications on crops that may be grown in multiple crop seasons per year. Seasonal application rates in most imidacloprid labels specify that these rates are equal to the yearly rates. For some crops, however, three application seasons/year are permitted. Three seasonal applications are permitted for the following crops groups: brassica, cucurbits, herbs & spices, leafy greens, leafy petiolates, legumes (except soybean), strawberries and tuberous corm vegetables. Since the current surface water model simulates application(s) for only one crop/year, it was assumed that the seasonal rate is equal to yearly rate for all the above-mentioned crop groups. This assumption means that calculated EECs for these crops were based on a significantly reduced rate (*i.e*., by one third) and consequently, the EECs for these crops are similarly reduced in situations where more than one crop is planted/year. Notably, the approach of multiplying the application rate by three was not chosen because EECs would not reflect actual application and growing practices. Furthermore, crop scenarios in the current model simulate plant growth for one crop season and one application/set of applications. Therefore, if three seasonal rates were distributed throughout the year, only one of them will be appropriately simulated with the crop growth scenario while the other two will be incorrectly simulated without the presence of the crop. Again, EECs generated by distributing the rate throughout the year will not accurately represent reality. A change in the label would be needed to match the assumption made in this assessment, otherwise EECs are significantly underestimated for crops with multiple growing seasons/year. By way of example, EECs resulting from use on herbs & spices was simulated using a research version of PWC which accepts applications to more than one season/year. EECs estimated from this simulation were higher by nearly five times.

### PFAM

Imidacloprid is used on cranberries and it is common practice to flood cranberry bogs. Water from cranberry bogs is generally released to adjacent waterbodies (wetlands, cannels, rivers, streams, lakes, or bays). The Pesticides in Flooded Applications Model (PFAM, version 2.0) was used to simulate cranberry agricultural practices.

PFAM was developed specifically for regulatory applications to estimate exposure for pesticides used in flooded agriculture such as cranberry bogs. The model considers the environmental fate properties of pesticides and allows for the specifications of common management practices that are associated with flooded agriculture such as scheduled water releases and refills. It estimates both acute and chronic concentrations over different durations and allows for defining different receiving water bodies.

PFAM was used to estimate the concentration of imidacloprid in water just before its release from a cranberry bog. The reported concentrations represent water introduced to the field (the bog) and not mixed with any additional water (*i.e.*, receiving water body). The concentration of imidacloprid is expected to be more than what would be expected in adjacent water bodies due to additional degradation and dilution unless the water is released into an empty canal, ditch, *etc.* The difference in the concentration of imidacloprid in the flood water to that in an adjacent waterbody depends on 1) the length of time imidacloprid is in the flooded bog, 2) the distance the water travels between the bog and the adjacent waterbody, 3) the amount of dilution, and 4) whether the flood water is mixed with additional water that also contains imidacloprid. PFAM can simulate application of a pesticide to a dry field and degradation in soil before water is introduced to the bog.

### Plant Assessment Tool (PAT)

The Plant Assessment Tool (PAT) is a mechanistic model that incorporates fate (*e.g.*, degradation) and transport (*e.g.,* runoff) data that are typically available for conventional pesticides, to estimate pesticide concentrations in terrestrial, wetland, and aquatic plant habitats. For terrestrial plants, runoff and erosion are modeled using PRZM and spray drift is modeled using AgDRIFT deposition values. The model uses a mixing cell approach to represent water within the active root zone area of soil, and accounts for flow through the terrestrial plant exposure zone (T-PEZ) caused by both treated field runoff and direct precipitation onto the T-PEZ. Pesticide losses from the T-PEZ occur from transport (*i.e*., washout and infiltration below the active root zone) and degradation. Wetlands are modeled using PRZM/VVWM and are then processed in PAT to estimate aquatic (mass per volume of water) and terrestrial (mass per area) concentrations. Aquatic plants exposure is modeled using the PRZM/VVWM models and the standard farm pond.

### Direct Water Applications

Direct application of imidacloprid to water is only labeled for granular and liquid formulations (Protector 2F, flowable and Protector 0.5G, granular) to control burrowing shrimp in commercial shellfish beds in Willapa Bay, WA. Imidacloprid labels, for this use, do not specify a target concentration but rather the rate is expressed in lbs. a.i/A (0.5 lbs. a.i/A). To calculate expected concentrations in the water following application, an estimate of the expected concentration of imidacloprid was calculated for the intertidal nearshore Bin 8 (Refer to ATTACHMENT 3-1 for characteristics/dimensions of Bin 8). The highest expected concentration in the near shore (Bin 8) is simply calculated, as shown below, from its volume and the amount of pesticide applied. Absorption to underlying sediment was not included in the calculation as it is expected to be low (low Koc for a sandy sediment).

*Water volume in one acre of Bin 8 = 4,046.86 m2(1 acre) x 0.5 meters = 2,023.43 m3 x 1,000 liters (1 m3) = 2,023,430 liters*

*Imidacloprid applied/Acre= 0.5 lbs. x 453.592 (1 lb.) = 226.796 grams x 1000 (1 gram) = 226,796 mg*

*Concentration = 226,796 mg/2,023,430 liters = 0.11209 mg/L or ppm = 112 ppb.*

### Poultry Litter Applications

In addition to traditional agricultural applications, imidacloprid can also be applied in poultry houses and livestock areas.

For poultry house use, the chicken litter collected from the poultry house applications could potentially be used as a soil amendment after it has been treated with imidacloprid. To assess the impacts of poultry litter use as a soil amendment, EPA modeled the amount of imidacloprid predicted to be in the poultry litter, as if it were applied to a corn field prior to planting. The poultry house use pattern evaluated by EPA represents an upper-end use pattern for products applied to poultry houses. The primary pest targeted by these products is the darkling beetle, which is mostly found on the perimeter portions of floors and lower walls, near feeders and water lines. While only portions of a poultry house may need to be treated, this is not explicitly stated or restricted on the product label. For modeling the highest exposure scenario, EPA conservatively assumed that the whole poultry house was treated each time a treatment is made. EPA assumed that six whole house treatments occurred per year, with a year being representative of the interval between complete, whole house litter clean outs. An application rate for imidacloprid-treated manure on a corn field was developed using the following process Shamblen and Judkins, 2012).

1. Application rate for (Credo® SC; EPA reg. No. 11556-146): 3 fl. oz of Credo/1000 ft2; treating a 20,000 ft2 house (maximum size poultry house) = 60 fl. oz Credo.
2. Credo contains 4.38 lbs. a.i/gallon; imidacloprid a.i. 60 oz Credo = 2.053125 lbs. imidacloprid.
3. A typical broiler house has six whole house treatments (6 flocks of broilers) before a full litter clean out, followed by storage, then application on a corn field. Treatment of 6 flocks results in application of 12.31875 lbs of imidacloprid (2.053125 x 6 lb a.i./application).
4. Six flocks will produce 168 tons of manure, and require 35 tons of bedding, resulting in a total of 203 tons of litter.
5. The cumulative residual concentration of imidacloprid in litter is 12.31875 lb/203 tons litter = 0.06 lb a.i./ton litter.
6. Maximum elemental nitrogen requirement for corn is 220 lb plant available nitrogen per acre (N/A)
7. Six flocks of broilers produce 14,400 lb nitrogen; 45% of this is assumed/estimated to be lost during storage, resulting in 7,920 lb of nitrogen.
8. Only 90% of the nitrogen is available to plants in the first year (USDA estimate of mineralization), resulting in 7,128 lb of plant available nitrogen.
9. An additional 50% of the nitrogen is lost during application, resulting in 3,564 lb plant-available nitrogen.
10. Based on the nitrogen application rate of 220 lbs. N/A, this results in 16.2 A being needed for the manure from six flocks (3,564 lb N/220 lb N/A = 16.2).
11. Based on a cumulative litter production of 203 tons, this results in a litter application rate of 12.5 tons/A (203 tons litter/16.2 A = 12.5 tons litter/A).
12. Based on a residual imidacloprid concentration in litter of 0.06 lb a.i./ton litter, and a litter application rate of 12.5 tons/A, the outdoor equivalent application rate for clothianidin is 0.7585 lb. a. i./A.

However, the maximum rate for imidacloprid to crops should not exceed 0.5 lb. a.i/A which is equivalent to an application of 8.3 tons of litter. Alternative poultry house treatment may occur, therefore a range of 8 to 0.6 tons of litter was assumed to be used in application to corn; equivalent to an application of 0.48 to 0.036 lb. imidacloprid a.i/A.

For livestock areas, the perimeters of the buildings are treated to reduce pests. Runoff from these treatments could potentially enter waterbodies and affect aquatic organisms. For these applications, EPA assumed a similar conceptual model to that used for residential applications (**Section 3.6**).

### Seed Treatment

Imidacloprid can also be applied as a seed treatment prior to the seed being planted. In many cases, the foliar and soil treatment applications can be made to the same crop and will generate much higher EECs than those from seed treatment applications. In some cases, foliar or soil applications are not permitted to a crop that can receive a seed treatment, such as corn seed. In these cases, because the application of treated poultry litter is assessed to corn fields (see previous section), EPA believes the EECs from this modeling is protective of corn seed treatment and the seed treatment only uses do not impact the overlap extent of the imidacloprid action area. As such, seed treatment applications were not modeled, but are discussed qualitatively in **APPENDIX 4-5**.

### Non-Agricultural Uses and Considerations

As described in the imidacloprid use matrix (**APPENDIX 1-2**) non-agricultural use patterns include: turf & ornamentals in nurseries and residential/commercial areas; poplar/cottonwood and Christmas tree plantations; poultry litter; tree injections; bait/ballets in farms/residential/commercial areas; pet collars and controlling burrowing shrimp in commercial shellfish beds in Willapa Bay, WA. Imidacloprid also has several indoor uses. As EFED does not have regionally specific information on indoor applications, EFED could not use typical down the drain models to assess EECs from indoor uses. As a result, EFED assumes that the EECs developed for the outdoor uses will be protective of the indoor uses.

Exposure from Urban, Suburban and Homeowner Uses

Imidacloprid has a number of registered uses that fit the general description of urban, suburban, or homeowner application. These include commercial/institutional/industrial premises, household/domestic dwellings, non-agricultural outdoor buildings, nuisance pests perimeter treatment, ornamental and/or shade trees, paths/patios, residential lawns, gardens, turf, along fences, porches *etc.*  When considered together, uses such as these could encompass a substantial fraction of an urban or suburban watershed. Table 3-6 summarizes these uses, rates, intervals for different areas.

Table 3-. Application Information for Modeled Homeowner Scenarios Based on Maximum Labeled Application Rates

|  |  |
| --- | --- |
| **PWC Scenario** | **Residential** |
| Included uses | Turf, ground covers, evergreens, flowering/foliage plants, foliage plants, roses, and small trees & shrubs and non-bearing fruits/nuts  House perimeter (insect and non-structural termite control, along fences, patios, garbage cans, under porches, shrubbery, firewood piles, ornamentals |
| Application rate | One Application of 0.40 lbs a.i./A1 |

1 Highest homeowner use rate Reg. No. 83851-7 allows for broadcast to lawns, recreational and ornamental turf areas to control various insect pests; control of nuisance pests in and around ornamentals and outside of buildings; and on home fruit and vegetable gardens.

EPA modeled residential uses employing the Residential ESA scenarios. EPA believes this approach is protective for all uses in a suburban/urban environment, as imidacloprid is expected to be mainly applied to pervious surfaces in these environments and the scenarios account for runoff from both pervious and impervious surfaces.

An estimate of the number of residential lots in a 10-ha watershed has been previously evaluated for California Red Legged Frog (CRLF) and other endangered species assessments [*i.e.*, **APPENDIX G** of “Potential Risks of Alachlor Use to Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Delta Smelt (*Hypomesus transpacificus*)”, USEPA 2009]. The assumption previously made was 58 lots arranged in 10 lot blocks (USEPA, 2009c). There are 10,890 ft2/lot x 58 lots in 10 ha = 631,620 ft2 out of a total of 1,076,391 ft2/ watershed (*i.e.*, 10 ha), resulting in an adjustment factor of 0.587. As a result, application rates for residential uses were adjusted by a factor of 0.587.

## Aquatic Modeling Input Parameters

The following sections discuss methods used for aquatic modeling. A summary of the environmental fate model input parameters used in the PWC for the modeling of imidacloprid is presented in **Table 3-7**. Input parameters were selected in accordance with the following EPA guidance documents: Guidance for *Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides*, Version 2.1[[3]](#footnote-4); *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[4]](#footnote-5)* (NAFTA, 2012; USEPA, 2012c), and *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[5]](#footnote-6).

Table 3-. Input Values Used for Tier II Surface Water Modeling with PWC or PFAM

| **Parameter (units)** | **Value (s)** | **Source (MRIDs)** | **Comments** |
| --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (Koc (L/kg)) | 266 | 425208-01; 420553-38 and 476994-44 | Average for 15 soils |
| Water Column Metabolism Half-life or Aerobic Aquatic Metabolism Half-life (days) 25°C | 236 | 484169-01 | The upper 90th confidence limit on the mean t½ from two values/systems |
| Benthic Metabolism Half-life or Anaerobic Aquatic Metabolism Half-life (days) 25°C | 81 | 422563- 78 | One whole-system t½ of 27 days multiplied by 3= 81 days |
| Aqueous Photolysis Half-life @ pH 7 (days) and Reference Latitude 40o N latitude, 25oC | 0.2 | 422563-76 |  |
| Hydrolysis Half-life (days) | Stable (0) | 490111-21 |  |
| Soil Half-life or Aerobic Soil Metabolism Half-life (days) 25°C | 254 | 420735-01; 452393-01/02/03; 498358-02 | The upper 90th confidence limit on the mean t½ from eight values for parent |
| Molecular Weight (M Wt. in g/mol) | 255.7 | Product Chemistry | |
| Vapor Pressure (VP in Torr) at 25oC | 1.5E-9 |
| Solubility in Water @ 25oC, pH not reported (mg/L) | 610 |
| Henry’s Law Constant (unitless) | 3.38E-11 | Calculated from VP, Solubility and M Wt. | |
| Foliar Half-life (days) | Stable (0) | Model default | |
| Application Efficiency | 0.99 (ground); 0.95 (air) | | |
| Drift | AgDRIFT, see **Section 3.4.2** | | |

## Aquatic Modeling Results

The estimated environmental concentrations (EECs) derived from the PWC modeling based on

maximum labeled rates, by HUC 2, are summarized for the various aquatic bins in **Table 3-8** and **Table 3-9** for water column and pore water, respectively. EECs for direct applications to cranberry are summarized in **Table 3-10**. The complete set of modeling inputs and results are available in **APPENDIX 3-1**. PWC runs and post-processed results are provided in **APPENDIX 3-2**. The range of EECs reflects the various application rates and timings for the various crops and scenarios modeled in the HUC 2 regions.

Table 3-8. Range of Daily Average Water Column EECs for Imidacloprid

| **HUC 2** | **Range of 1-in-15 year Daily Average EECs (µg/L)** | | | | | | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Bin 1** | **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| HUC 1 | 2.16 - 40.4 | 4 - 85 | 1.26 - 24.28 | 1.26 - 24.28 | 4 - 85 | 0.76 - 17.01 | 0.76 - 17.01 |
| HUC 2 | 1.26 - 34.7 | 3 - 117 | 0.66 - 24.57 | 0.66 - 24.57 | 3 - 117 | 0.44 - 14.27 | 0.44 - 14.27 |
| HUC 3 | 1.69 - 47.9 | 4 - 97 | 0.84 - 26.6 | 0.84 - 26.6 | 4 - 97 | 0.52 - 17.19 | 0.52 - 17.19 |
| HUC 4 | 2.14 - 56.8 | 4 - 148 | 0.73 - 16.95 | 0.73 - 16.95 | 4 - 148 | 0.38 - 9.63 | 0.38 - 9.63 |
| HUC 5 | 2.2 - 39.8 | 7 - 97 | 1.10 - 20.51 | 1.10 - 20.51 | 7 - 97 | 0.66- 13.08 | 0.66- 13.08 |
| HUC 6 | 1.42 - 45.2 | 4 - 104 | 0.52 - 21.38 | 0.52 - 21.38 | 4 - 104 | 0.26 - 14.13 | 0.26 - 14.13 |
| HUC 7 | 1.86 - 53.6 | 5 - 126 | 1.075 - 25.06 | 1.075 - 25.06 | 5 - 126 | 0.81 - 12.84 | 0.81 - 12.84 |
| HUC 8 | 1.88 - 54.1 | 7 - 158 | 1.38- 32.83 | 1.38- 32.83 | 7 - 158 | 1.19 - 22.16 | 1.19 - 22.16 |
| HUC 9 | 1.16 - 135 | 3 - 57 | 0.46 - 14.84 | 0.46 - 14.84 | 3 - 57 | 0.24 - 8.88 | 0.24 - 8.88 |
| HUC 10a | 3.08 - 87.3 | 6 - 86 | 1.42 - 19.15 | 1.42 - 19.15 | 6 - 86 | 0.76 - 10.25 | 0.76 - 10.25 |
| HUC 10b | 2.35 - 146 | 6 - 88 | 0.86 - 11.55 | 0.86 - 11.55 | 6 - 88 | 0.44 - 5.91 | 0.44 - 5.91 |
| HUC 11a | 2.79 - 65.9 | 6 - 152 | 1.51- 34.25 | 1.51- 34.25 | 6 - 152 | 0.88 - 20.76 | 0.88 - 20.76 |
| HUC 11b | 3.29 - 70.8 | 6 - 153 | 1.41 - 19.49 | 1.41 - 19.49 | 6 - 153 | 0.74 - 11.54 | 0.74 - 11.54 |
| HUC 12a | 2.43 - 70.9 | 8 - 150 | 0.95 - 31.39 | 0.95 - 31.39 | 8 - 150 | 0.53 - 18.74 | 0.53 - 18.74 |
| HUC 12b | 2.69 - 69.2 | 7 - 154 | 0.68 - 25.62 | 0.68 - 25.62 | 7 - 154 | 0.38 - 20.26 | 0.38 - 20.26 |
| HUC 13 | 1.74 - 46 | 8 - 150 | 0.42 - 12.59 | 0.42 - 12.59 | 8 - 150 | 0.19 - 6.22 | 0.19 - 6.22 |
| HUC 14 | 3.58 - 122 | 8 - 173 | 0.46 - 13.75 | 0.46 - 13.75 | 8 - 173 | 0.20 - 6.51 | 0.20 - 6.51 |
| HUC 15a | 3.21 - 94.1 | 8 - 224 | 1.13 - 35.63 | 1.13 - 35.63 | 8 - 224 | 0.55 - 19.97 | 0.55 - 19.97 |
| HUC 15b | 2.58 - 86.4 | 8 - 228 | 0.53 - 26.34 | 0.53 - 26.34 | 8 - 228 | 0.26 - 13.1 | 0.26 - 13.1 |
| HUC 16a | 2.08 - 96.3 | 8 - 180 | 0.49 - 14.63 | 0.49 - 14.63 | 8 - 180 | 0.22 - 7.05 | 0.22 - 7.05 |
| HUC 16b | 1.47 - 100 | 8 - 184 | 0.16- 6.32 | 0.16- 6.32 | 8 - 184 | 0.07 - 2.88 | 0.07 - 2.88 |
| HUC 17a | 1.92 - 46.3 | 6 - 96 | 0.63 - 24.77 | 0.63 - 24.77 | 6 - 96 | 0.28 - 15.95 | 0.28 - 15.95 |
| HUC 17b | 1.47 - 88.8 | 5 - 91 | 0.24 - 10.05 | 0.24 - 10.05 | 5 - 91 | 0.10 - 5.67 | 0.10 - 5.67 |
| HUC 18a | 1.53 - 61.5 | 8 - 161 | 0.61 - 16.8 | 0.61 - 16.8 | 8 - 161 | 0.42 - 8.03 | 0.42 - 8.03 |
| HUC 18b | 1.79 - 50.2 | 8 - 151 | 0.48 - 15.89 | 0.48 - 15.89 | 8 - 151 | 0.23 - 7.92 | 0.23 - 7.92 |
| HUC 19a | 1.97 - 81.2 | 5 - 96 | 0.59- 10.51 | 0.59- 10.51 | 5 - 96 | 0.28- 5.84 | 0.28- 5.84 |
| HUC 19b | 1.7 - 46.9 | 5 - 96 | 0.58 - 25.44 | 0.58 - 25.44 | 5 - 96 | 0.30 - 14.77 | 0.30 - 14.77 |
| HUC 20a | 4.03 - 68.1 | 9 - 163 | 3.36 - 33.97 | 3.36 - 33.97 | 9 - 163 | 2.97 - 28.27 | 2.97 - 28.27 |
| HUC 20b | 5.84 - 62.4 | 8 - 152 | 2.89 - 39.32 | 2.89 - 39.32 | 8 - 152 | 1.40 - 35.52 | 1.40 - 35.52 |
| HUC 21 | 5.85 - 54 | 8 - 155 | 5.17 - 38.07 | 5.167 - 38.07 | 8 - 155 | 3.51 - 30.59 | 3.51 - 30.59 |

Table 3-9. Range of Pore Water EECs for Imidacloprid

| **HUC 2** | **Range of 1-in-15 year Daily Average EECs (µg/L)** | | | | | |
| --- | --- | --- | --- | --- | --- | --- |
| **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| HUC 1 | 4 - 111 | 0.75 - 72.04 | 0.75 - 72.04 | 4 - 111 | 0.40 - 17.28 | 0.40 - 17.28 |
| HUC 2 | 5 - 151 | 0.76 - 52.06 | 0.76 - 52.06 | 5 - 151 | 0.21- 12.52 | 0.21- 12.52 |
| HUC 3 | 4 - 98 | 0.26 - 11.1 | 0.26 - 11.1 | 4 - 98 | 0.13 - 5.04 | 0.13 - 5.04 |
| HUC 4 | 4 - 148 | 0.23 - 11.45 | 0.23 - 11.45 | 4 - 148 | 0.11 - 2.46 | 0.11 - 2.46 |
| HUC 5 | 9 - 126 | 0.72 - 16.97 | 0.72 - 16.97 | 9 - 126 | 0.28 - 4.40 | 0.28 - 4.40 |
| HUC 6 | 4 - 104 | 0.15 - 8.81 | 0.15 - 8.81 | 4 - 104 | 0.06- 3.93 | 0.06- 3.93 |
| HUC 7 | 7 - 135 | 0.78 - 25.93 | 0.78 - 25.93 | 7 - 135 | 0.41- 5.59 | 0.41- 5.59 |
| HUC 8 | 9 - 158 | 1.03 - 225.4 | 1.03 - 225.4 | 9 - 158 | 0.70 - 59.58 | 0.70 - 59.58 |
| HUC 9 | 3 - 67 | 0.14 - 5.88 | 0.14 - 5.88 | 3 - 67 | 0.07 - 2.80 | 0.07 - 2.80 |
| HUC 10a | 7 - 89 | 0.46 - 6.19 | 0.46 - 6.19 | 7 - 89 | 0.21 - 2.86 | 0.21 - 2.86 |
| HUC 10b | 7 - 93 | 0.24 - 3.65 | 0.24 - 3.65 | 7 - 93 | 0.10 - 1.62 | 0.10 - 1.62 |
| HUC 11a | 7 - 158 | 0.58 - 16.17 | 0.58 - 16.17 | 7 - 158 | 0.25 - 6.85 | 0.25 - 6.85 |
| HUC 11b | 7 - 158 | 0.46 - 6.23 | 0.46 - 6.23 | 7 - 158 | 0.19- 3.13 | 0.19- 3.13 |
| HUC 12a | 8 - 156 | 0.35 - 13.97 | 0.35 - 13.97 | 8 - 156 | 0.15 - 6.20 | 0.15 - 6.20 |
| HUC 12b | 7 - 160 | 0.20 - 14.69 | 0.20 - 14.69 | 7 - 160 | 0.10 - 6.62 | 0.10 - 6.62 |
| HUC 13 | 8 - 156 | 0.13 - 2.82 | 0.13 - 2.82 | 8 - 156 | 0.04 - 1.22 | 0.04 - 1.22 |
| HUC 14 | 8 - 183 | 0.18 - 3.86 | 0.18 - 3.86 | 8 - 183 | 0.07- 1.62 | 0.07- 1.62 |
| HUC 15a | 8 - 227 | 0.30 - 23.63 | 0.30 - 23.63 | 8 - 227 | 0.13 - 8.45 | 0.13 - 8.45 |
| HUC 15b | 8 - 231 | 0.15 - 9.63 | 0.15 - 9.63 | 8 - 231 | 0.05 - 4.02 | 0.05 - 4.02 |
| HUC 16a | 8 - 188 | 0.22 - 3.92 | 0.22 - 3.92 | 8 - 188 | 0.07 - 1.83 | 0.07 - 1.83 |
| HUC 16b | 8 - 193 | 0.05 - 2.08 | 0.05 - 2.08 | 8 - 193 | 0.02 - 0.88 | 0.02 - 0.88 |
| HUC 17a | 6 - 96 | 0.19 - 8.28 | 0.19 - 8.28 | 6 - 96 | 0.07 - 4.87 | 0.07 - 4.87 |
| HUC 17b | 6 - 94 | 0.07 - 4.84 | 0.07 - 4.84 | 6 - 94 | 0.03 - 2.53 | 0.03 - 2.53 |
| HUC 18a | 8 - 164 | 0.30 - 13.49 | 0.30 - 13.49 | 8 - 164 | 0.10 - 4.47 | 0.10 - 4.47 |
| HUC 18b | 9 - 154 | 0.13 - 5.60 | 0.13 - 5.60 | 9 - 154 | 0.05 - 2.48 | 0.05 - 2.48 |
| HUC 19a | 5 - 99 | 0.20 - 5.76 | 0.20 - 5.76 | 5 - 99 | 0.09 - 2.76 | 0.09 - 2.76 |
| HUC 19 b | 5 - 100 | 0.20 - 12.37 | 0.20 - 12.37 | 5 - 100 | 0.12 - 11.08 | 0.12 - 11.08 |
| HUC 20a | 9 - 164 | 1.06 - 9.87 | 1.06 - 9.87 | 9 - 164 | 0.10 - 7.11 | 0.10 - 7.11 |
| HUC 20b | 8 - 153 | 0.68 - 8.16 | 0.68 - 8.16 | 8 - 153 | 0.34 - 7.25 | 0.34 - 7.25 |
| HUC 21 | 9 - 157 | 0.78 - 19.25 | 0.78 - 19.25 | 9 - 157 | 0.73- 5.88 | 0.73- 5.88 |

Table 3-10. Imidacloprid EECs for Use on Cranberry Bogs

|  |  |  |  |
| --- | --- | --- | --- |
| **Scenario Application Date (Rate Kg a.i./Hac)1** | **HUC 2** | **State** | **Daily Average EECs (µg/L)** |
| MA\_Cranberry\_Winter Flood\_SCH (0.561) | 01 | MA, ME, CT, RI, NH, VT | 188 |
| MA\_Cranberry\_Winter Flood\_SCH (0.561) | 02 | NJ, NY, DE, PA, VT | 188 |
| WI\_Cranberry\_Winter Flood\_SCH (0.561) | 04 | WI, MI, NY | 191 |
| WI\_Cranberry\_Winter Flood\_SCH (0.561) | 05 | NY, PA | 191 |
| WI\_Cranberry\_Winter Flood\_SCH (0.561) | 07 | WI, MO | 191 |
| WI\_Cranberry\_Winter Flood\_SCH (0.561) | 09 | MN | 191 |
| OR\_Cranberry\_No Flood\_SCH (0.561) | 17a and b | WA, OR | 2.76 |
| OR\_Cranberry\_Winter Flood\_SCH (0.561) | 17a and b | WA, OR | 150 |
| OR\_Cranberry\_No Flood\_SCH (0.561) | 18a and b | OR | 2.76 |
| OR\_Cranberry\_Winter Flood\_SCH (0.561) | 18a and b | OR | 150 |
| **1**One soil application, pre-bloom (mid-May) applied by chemigation and soil surface band spray followed by irrigation. | | | |

## Available Monitoring Data

Examination of the EPA 303(d) list of impaired waters[[6]](#footnote-7) on May 11, 2021, indicates no impairments caused by imidacloprid.

Water monitoring data were obtained from the National Water Quality Monitoring Council’s Water Quality Data Portal (USEPA and USGS, 2013), which is supplied by the USGS NWIS and EPA STORET databases of monitoring data collected across the United States by numerous federal, state, tribal and local agencies. Prominent contributors to this database include the USGS National Water-Quality Assessment Program (NAWQA) and various state and local agencies.

In general, the surface water monitoring data include sampling sites that represent a range of aquatic environments including small and large water bodies, rivers, reservoirs, and urban and agricultural locations, but are limited for some areas of the United States where imidacloprid use occurs. Also, the sampling sites, as well as the number of samples, vary by year. The vulnerability of the sampling site to imidacloprid contamination varies substantially due to use, soil characteristics, weather and agronomic practices. Often, monitoring programs in the Water Quality Portal are not specifically designed to target imidacloprid use; as such, peak concentrations of imidacloprid likely went undetected in these programs. Overall, the extent to which historical values represent current agronomic or labeled use instructions is uncertain.

While there are many individual samples collected and analyzed for imidacloprid across the United States, it would not be appropriate to combine these data sources to generate exposure estimates or to use these datasets to represent exposure on a national or even regional basis. While these data demonstrate potential exposure, using the measured concentrations as an upper bound exposure estimate would not be a reasonable approach for the reasons given above, including limited sample frequency, limited use information, and sampling site variability, on a national or even a regional basis. Therefore, model estimated concentrations should be considered a suitable upper bound concentration for imidacloprid.

### Water Quality Portal

Comprehensive surface water and groundwater imidacloprid data were obtained in March 2021 in a download of data from the Water Quality Data Portal ([http://www.waterqualitydata.us](http://www.waterqualitydata.us/)/). **Table 3-11** provides a summary of the results by HUC 2 region, with sampling occurring from 1999 to 2021.

Table 3-11. Water Quality Portal Monitoring Data Summarized by 2-digit HUC for Imidacloprid

| **HUC-21** | **Years** | **Number of Sites** | **Number of Samples** | | **Sample Detection Frequency** | **Measured Detection Range (µg/L)** |
| --- | --- | --- | --- | --- | --- | --- |
| **Total** | **With No Detections** |
| 1 | 2001 - 2020 | 61 | 839 | 503 | 40% | 6.80E-03 - 1.23E+00 |
| 2 | 1999 - 2020 | 416 | 2,813 | 2,157 | 23% | 2.00E-03 - 7.94E+00 |
| 3 | 1999 - 2021 | 2,263 | 9,711 | 5,349 | 45% | 1.60E-03 - 1.30E+01 |
| 4 | 2000 - 2020 | 263 | 1,338 | 913 | 32% | 3.80E-03 - 9.85E-01 |
| 5 | 1999 - 2020 | 49 | 1,216 | 689 | 43% | 5.20E-03 - 5.09E-01 |
| 6 | 1999 - 2020 | 16 | 242 | 233 | 4% | 7.00E-03 - 2.00E-01 |
| 7 | 2000 - 2020 | 468 | 4,310 | 3,610 | 16% | 3.80E-03 - 2.23E+00 |
| 8 | 1999 - 2020 | 38 | 1,213 | 605 | 50% | 6.80E-03 - 2.27E+00 |
| 9 | 2001 - 2020 | 124 | 530 | 502 | 5% | 6.80E-03 - 4.30E-02 |
| 10 | 1999 - 2020 | 159 | 2,344 | 1,800 | 23% | 1.80E-03 - 2.15E+00 |
| 11 | 1999 - 2020 | 46 | 860 | 551 | 36% | 6.80E-03 - 3.17E+00 |
| 12 | 2001 - 2020 | 57 | 1,167 | 763 | 35% | 6.80E-03 - 4.60E+00 |
| 13 | 2004 - 2020 | 30 | 359 | 348 | 3% | 6.80E-03 - 3.69E-01 |
| 14 | 2000 - 2020 | 67 | 546 | 539 | 1% | 7.37E-03 - 1.68E-01 |
| 15 | 2001 - 2020 | 69 | 455 | 448 | 2% | 2.49E-03 - 1.30E-01 |
| 16 | 1999 - 2020 | 31 | 574 | 463 | 19% | 6.80E-03 - 1.24E-01 |
| 17 | 1999 - 2020 | 708 | 8,997 | 8,150 | 9% | 4.80E-05 - 4.49E+00 |
| 18 | 2000 - 2020 | 433 | 2,284 | 1,676 | 27% | 2.00E-03 - 4.18E+00 |
| 19 | 2013 - 2019 | 1 | 20 | 20 | 0% | NA |
| 20 | 1999 - 2019 | 46 | 76 | 63 | 17% | 6.11E-03 - 4.94E+00 |
| 21 | 2013 | 2 | 2 | 2 | 0% | NA |

## Open Literature Monitoring Data

More targeted sampling has been conducted by various independent study authors with results available in the open literature. A full summary of open literature monitoring data as of 2017 is available in **APPENDIX 3-3**. Maximum monitored values can be higher but mostly within an order of magnitude of values reported from the Water Quality Portal.

## Aquatic Exposure Summary

Model-derived EECs represent an upper bound on potential exposure resulting from imidacloprid use. Comparing the concentrations in the medium and high flowing and static bins (3, 4, 6, and 7) to the highest measured concentrations, the modeled values are much higher than the measured concentrations. The medium and high flowing and static bins were used for comparison purposes as these would seem to represent typical waterbodies considered for ambient water monitoring, as they typically have flow and water present all year. As recommended by the NRC in the 2013 NAS report, general monitoring data are not recommended to be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of EPA’s fate and transport models. However, EPA believes monitoring data can be used as part of the weight-of-evidence evaluation to evaluate potential exposure.

## Uncertainties in Aquatic Modeling and Monitoring Estimates

Exposure to aquatic organisms from pesticide applications is estimated using PWC EECs. Regional differences in exposure are assessed using regionally specific PWC scenarios (*e.g*., information on crop growth and soil conditions) and meteorological conditions at the HUC 2 level (**Section 3.3. Scenario Selection**). The information used in these scenarios is designed to reflect conditions conducive to runoff. In instances where PWC scenarios do not exist in a HUC 2, surrogate scenarios from other HUCs are used. For fields where agricultural practices that result in less conservative scenario parameters are employed (*i.e*., conditions less conducive to runoff and pesticide loading of waterbodies), the potential for lower EECs would be expected.

The static waterbodies modeled with PWC are fixed volume systems with no outlet, resulting in the potential for accumulation of pesticide over time. Effects due to the increase and/or decrease of the water level in the waterbody and thus the concentration of pesticide in the waterbody are not modeled.

Flowing waterbodies are modeled in the PWC using the constant volume and flow through custom waterbody option. Effects due to the increase and/or decrease of the water level and flowrate in the waterbody and thus the concentration of pesticide in the waterbody are not modeled.

The assessment relies on maximum use patterns (**APPENDIX 1-2**). In situations where use patterns are less than the labeled maximums, environmental exposures will be lower.

The aquatic modeling conservatively assumes that the waterbody abuts the treated area. As such, any reduction in loading from runoff that could occur as the result of managed vegetative filter strips or unmanaged naturally occurring interfaces between treated areas and waterbodies are not considered.

The aquatic modeling assumes a constant wind of 10 mph blowing directly toward the waterbody (**Section 3.4.2. Spray Drift**). These assumptions are conducive to drift transport and result in maximum potential loading to the waterbody. However, in many situations the wind will not be blowing constantly and directly toward the water body at this speed; therefore, aquatic deposition will likely be less than predicted. Additionally, many labels and applicator best management practices encourage not applying pesticides when the wind is blowing in the direction of sensitive areas (*i.e.*, listed species habitat). Lastly, reductions in spray drift deposition due to air turbulence, interception of spray drift on nearby plant canopy, and applications during low wind speeds are not considered in the spray drift estimates; therefore, loading due to spray drift may be over-estimated.

There is uncertainty associated with the selection of PWC input parameters. In this regard, one of the important parameters that can impact concentration estimates is the selection of application dates (**Section 3.4.3. Application Timing**); runoff and potential pesticide loading are greatest when applications immediately precede major precipitation events. Although the pesticide application dates are selected to be appropriate and protective (*i.e.*, selected with consideration for label restrictions and simulated cropping dates, pest pressures, and high precipitation meteorological conditions), uncertainty nevertheless results because the application window (the time span during a season that a pesticide may likely be applied) for a pesticide may be wide and actual application dates may vary over the landscape. While data sources exist that allow for determination of historical application dates (*e.g.*, California’s Pesticide Use Report and pesticide use surveys), it is uncertain how these dates reflect future application events. Additionally, the PWC model uses the same application dates for the 30-year simulation. While it is unlikely that an application would occur on the same dates every year for 30 years, this modeling process allows for a distribution of EECs to be developed that captures the peak loading events.

The PWC is a field-scale model. Flowing water bodies such as streams and rivers with physical parameters consistent with aquatic bins 3 and 4 have watershed areas well beyond those of typical agricultural fields. Initial modeling efforts in previous BEs, applying the field-scale model to these large watersheds and using the same scenario parameters as those used for the other bins, resulted in extremely high EECs which have not been observed in the environment, nor would be expected to occur due to fluid dynamic processes such as advective dispersion **(Figure 3-2**), where the peak concentration is dampened as it moves from a low flowing stream (bin 2) to a higher flowing river (bins 3 and 4). It is acknowledged that a watershed/basin-scale model capable of evaluating the impact of pesticide and water transport at the field-scale and aggregating these loadings to waterbodies at the larger watershed-scale is needed to evaluate these flowing aquatic systems.

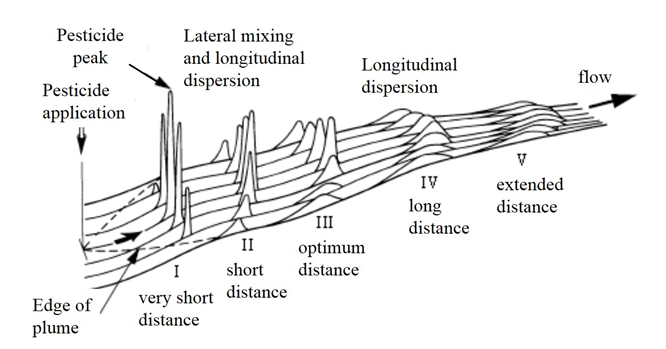


Figure 3-2. Effect of Pesticide Concentration via Advective Dispersion

## Uncertainties the Plant Assessment Tool (PAT)

The PAT model does not account for site specific field management and hydrology (*e.g*., terracing, contour farming, runoff and erosion controls, irrigation/drainage ditches, rills and creeks) which may result in less opportunity for runoff into the T-PEZ. Many different factors (*e.g*., slope; surface roughness; flow path length) can influence the occurrence, distance of, and prevalence of runoff onto the T-PEZ. These factors may vary greatly between different application sites (*e.g.*, corn; wheat; potato; grape; bare field; turf).

The PAT model assumes that the water leaving the field as surface runoff is driven primarily by the amount of rainfall and the curve number, which is a function of the land use (*i.e*., row crops, pasture, fallow), management (*i.e*., straight row cropping, conservation tillage, *etc.*), and hydrologic soil conditions (*i.e*., high runoff potential with very slow infiltration rates; Young and Fry, 2016). Runoff leaving the field is assumed to enter the T-PEZ along the downslope field edge and coverage of the T-PEZ area conceptually happens instantaneously as the calculations are on a daily timestep rather than shorter timestep (*e.g*., hourly). As a result, the T-PEZ does not account for differences in the runoff loading (*e.g.*, point entry and fan shaped sheet flow *vs.* uniform sheet flow entry), gradients in concentration due to interception and infiltration (*e.g.*, buffering capacity of the T-PEZ), rain intensity and infiltration capacity relationships (*e.g*., pulsed rain events vs. one intense rain event). These natural features of the landscape may result in higher concentrations from runoff at the edge of the T-PEZ nearer the treated field than estimated in the model.

There are many different types of wetlands (*e.g*., depressional, groundwater fed, flow through, permanently flooded, and ephemeral) that may be present in landscapes receiving runoff from pesticide use sites. The default WPEZ model was selected to be representative and conservative (in terms of final pesticide concentrations) and acts as a surrogate for other types of wetlands. This assumption may result in overestimation of pesticide loading and fate than would be observed in some wetland systems.

# Measures of Terrestrial Exposure

Terrestrial animals may be exposed to imidacloprid through multiple routes of exposure, including diet, drinking water, dermal and inhalation exposure. Terrestrial dietary items may consist of plants, invertebrates or vertebrates (amphibians, reptiles, birds or mammals) that inhabit terrestrial areas or aquatic dietary items (fish, invertebrates or plants). However, due to imidacloprid’s log Kow value (0.57), significant bioaccumulation in aquatic food items is not expected and potential risk from this route of exposure is considered low. Therefore, estimates of exposure through consumption of aquatic food items using KABAM or BCF values are not calculated. A detailed discussion of the conceptual framework for estimating terrestrial exposure concentrations is provided in **ATTACHMENT 1-1**.

Two major parameters are used in terrestrial exposure modeling to characterize a species: body weight and diet. Estimates of body weights are necessary to estimate dose-based exposures through diet, drinking water, inhalation and dermal exposure routes. Information on the dietary requirements of listed species are necessary to determine relevant exposures through consumption of contaminated prey. Species-specific assumptions related to diet and body weight are provided within the model. The foliar dissipation half-life of the chemical can also impact the duration of exposure to predicted terrestrial EECs. EFED’s default foliar dissipation rate of 35 days was used for this analysis to estimate dissipation after each application.

To improve efficiency and expand EFED’s modeling capabilities to other, non-dietary routes of exposure for terrestrial organisms, models have been developed to integrate the relevant exposure pathways and allow for batch processing of multiple analyses. For use in the pilot BEs (USEPA, 2016f), the Terrestrial Effects Determination (TED) tool was developed, which integrated T-REX, T-HERPS, the earthworm fugacity model, components of KABAM and AgDRIFT into one model platform. As part of the development of the draft biological opinion for chlorpyrifos, diazinon and malathion, the TED tool was converted to the terrestrial MAGtool, which further expanded the TED tool to predict the magnitude of effect at a population scale and incorporate the degree of overlap of a species range with potential use sites for a chemical (and associated off site transport areas) into the effects determination. The MAGtool has replaced the TED tool for modeling terrestrial exposure in the biological evaluations. A complete description of the MAGtool can be found in **ATTACHMENT 4-1**.

When the MAGtool is run for each species, terrestrial exposure concentrations are uniquely calculated for each species depending on relevant use overlap with the species range, available usage data, application rates associated with these relevant uses and the dietary items, habitat and obligate relationships for that species. As EECs will vary for each species, they are reported with the individual species in the individual effects determinations (**APPENDIX 4-1**).

To provide a bounding of potential terrestrial EECs used in the effects determinations, EECS were calculated for the range of application rates for imidacloprid (a lower bound application rate of 0.045 lb a.i./A with 1 application per year and an upper bound application rate of 0.5 lb a.i./A with 1 application per year) and are provided below in **Table 3-**. How the EECs will be applied will vary with each step of the analysis (*e.g.,* use of upper bound EECs in Step 1 vs. distribution of EECs in Step 2) and could be slightly higher with mid-range application rates applied multiple times. Additionally, other information considered in Step 2 (*e.g.,* typical use rates, use rates based on maximum usage in a species range, distribution of EECs, etc.), could alter the EECs used to assess a species exposure. All uses for imidacloprid and associated application rates are provided in **APPENDIX 1-2**. **Table 3-** summarizes the mean and upper bound dietary-based EECs and the associated base model that is used in the MAGtool to predict the EECs. Imidacloprid uses also include granular and seed formulations; these are analyzed separately and are discussed in **APPENDIX 4-5**.

As discussed above, the T-REX model default EECs were used to estimate mean and upper bound residues on arthropods and plant tissues. These default EECs are based in part on a historical database of residue measurements made for a variety of pesticides after foliar applications to different crops (Fletcher *et al.*, 1994). However, for the neonicotinoids, chemical-specific data have also been submitted to the Agency based on field residue studies that quantify the concentrations of these active ingredients in different plant tissues (*e.g.*, pollen, nectar, flower, leaf) following foliar, soil, and/or seed applications. Therefore, in order to further characterize the T-REX default EECs and their associated level of protection for applicable terrestrial listed species, an analysis was conducted to compare neonicotinoid residues measured in leaves after foliar applications to the T-REX default EECs for broadleaf plants (**ATTACHMENT 3-2**). This analysis also compared neonicotinoid residues measured in leaves following soil applications to those following foliar applications (and to the default T-REX EECs).

This analysis demonstrated that following foliar applications of neonicotinoids, very few (2.5%) of the daily average residue concentrations measured in leaves from registrant-submitted field residue studies exceeded the T-REX upper bound estimated environmental concentration (EEC) for broadleaf plants when normalized to a common application rate. Additionally, 12% of the daily average residue concentrations measured in leaves exceeded the T-REX mean EEC for broadleaf plants. These few exceedances of the T-REX EECs typically occurred soon after application (within the first few days), after which residues dissipated rapidly relative to the default T-REX EECs which assumed a dissipation half-life of 35 days. Following soil applications, neonicotinoid residues in leaves never exceeded the mean or upper bound T-REX EECs in broadleaf plants. However, residues associated with soil applications did not show the same rapid decline over time compared to those associated with foliar applications, likely due to the continued uptake of the chemical from the soil over time. In general, these comparisons indicate that the T-REX EECs are appropriately conservative in that very few measured residue values exceeded the T-REX default EECs while the vast majority were well below the default EECs.

Table 3-12. Mean and Upper-Bound Dietary Based EECs Calculated for Food Items Consumed by Listed Mammals, Birds, Terrestrial-phase Amphibians or Reptiles Based on Foliar Applications.

Values represent potential exposures for animals feeding on the treated field or in adjacent habitat directly adjacent to the field.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Food Item** | **Base Model** | **Lower bound application rate**  **(0.045 lb a.i./A x 1 application/year)** | | **Upper bound application rate**  **(0.5 lb a.i./A x 1 applications/year)** | |
| **Upper Bound** | **Mean** | **Upper Bound** | **Mean** |
| Short Grass | T-REX | 10.8 | 3.8 | 120 | 42.5 |
| Tall Grass, nectar and pollen | T-REX | 5.0 | 1.6 | 55 | 18 |
| Broadleaf plants | T-REX | 6.1 | 2.0 | 67.5 | 22.5 |
| Seeds, fruit and pods | T-REX | 0.68 | 0.32 | 7.5 | 3.5 |
| Arthropods (above ground) | T-REX | 4.2 | 2.9 | 47 | 32.5 |
| Soil-dwelling invertebrates (earthworms) | Earthworm fugacity | 0.0011 | NA | 0.012 | NA |
| Small mammals (15 g, short grass diet) | T-HERPS | 10.98 | 3.89 | 122 | 43.2 |
| Large mammals (1000 g, short grass diet)2 | T-HERPS | 1.76 | 0.62 | 19.6 | 6.92 |
| Small birds (20 g, insect diet) | T-HERPS | 9.2 | 6.37 | 102 | 70.7 |
| Small terrestrial phase amphibians or reptiles (2 g; insect diet) | T-HERPS | 0.45 | 0.31 | 4.99 | 3.45 |
| Fish, aquatic invertebrates and aquatic plants | NA3 |  | | | |

1 NA as upper bound and mean residues only applicable to items dependent on residues on foliage

2 Also represent residues in carrion.

3 NA due to lack of bioaccumulation of imidacloprid in aquatic dietary items

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6. [https://](https://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T)[Specific State Causes of Impairment | Water Quality Assessment and TMDL Information | US EPA](https://iaspub.epa.gov/waters10/attains_nation_cy.cause_detail_303d?p_cause_group_id=885) [↑](#footnote-ref-7)