Chapter 3 – Thiamethoxam Exposure Characterization

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# Environmental Transport and Fate Characterization

Thiamethoxam is soluble (4100 mg/L) in water. The vapor pressure (4.95 x 10-11 mm Hg) and Henry's Law constants (4.63 x 10-15 atm·m3/mol) indicate that the compound is relatively non-volatile under field conditions. The compound does not dissociate within the range of pH 2 to 12. The n-octanol water partition coefficient (log Kow = -0.13) indicates a low potential for bioaccumulation. General physical and chemical properties of thiamethoxam are summarized in **Table 3-1**.

The main routes of dissipation from a treated site are spray drift, runoff, microbial degradation under aerobic and anaerobic aquatic conditions and aqueous photolysis. Thiamethoxam is expected to reach surface water primarily through spray drift and transport through runoff of the dissolved phase of thiamethoxam.

Table 3-1. Physical and Chemical Properties of Thiamethoxam

|  |  |
| --- | --- |
| **Property** | **Value** |
| Molecular Weight (g/mol) | 291.7 |
| Water Solubility @ 25°C (mg/L) | 4100 |
| Vapor Pressure@ 25°C (torr or mm Hg) | 4.95 x 10-11 |
| Henry’s Law Constant (atm·m3/mol)1 | 4.63 x 10-15 |
| Kow(log Kow) | 0.74 (-0.13) |
| Dissociation Constant | no dissociation from pH 2-12 |
| 1 = Henry’s Law Constant is calculated from (VAPR/760)/(SOL/MWT), where VAPR is vapor pressure in torr, MWT is molecular weight in g/mol, and SOL is the solubility in water in mg/L. | |

*Degradation and Metabolism*

In terrestrial environments, thiamethoxam is expected to be persistent, with half-lives on the order of months to years. Thiamethoxam persists from months to years in various aerobic soils with (14) half-lives ranging from 34.3 to 464 days. Thiamethoxam persists for months with anaerobic soil half-lives ranging from 45.6 to 118 days from two anaerobic soil metabolism studies. Photodegradation in soil is not expected to be a substantial route of dissipation, as half-lives range from 80 to 97 days in irradiated soil.

Thiamethoxam is less persistent in aquatic environments, with half-lives on the order of weeks. In aerobic aquatic metabolism studies, thiamethoxam degraded with half-lives ranging from 16.2 to 35.1 days in sediment water systems. Thiamethoxam showed similar persistence in anaerobic aquatic environments with half-lives ranging 20.7 to 28.6 days. In clear, acidic waters, thiamethoxam is expected to be less persistent, as photodegradation in water (3.4-3.9 d) and alkaline-catalyzed hydrolysis (4.2-8.4 d) half-lives are on the order of days.

*Soil Sorption and Mobility*

Batch equilibrium studies indicate that thiamethoxam is mobile to moderately mobile in soils according to the FAO mobility classification (FAO, 2000). The adsorption, Koc ranged from 33.1-176.7 mL/goc. The study results indicate correlation between thiamethoxam adsorption to soil and percent organic carbon. No correlation is found between thiamethoxam adsorption and percent clay. The desorption Koc values were higher than the adsorption Koc values, indicating that once adsorbed to soil, thiamethoxam would be less likely to be mobile in soil. Aged leaching studies also suggest that thiamethoxam becomes less mobile after aging. This data supports unextracted residues (11-59% AR) will most likely bind to soil and sediment. Given these lines of evidence, in addition to the fact that exhaustive extraction techniques were utilized to extract thiamethoxam, unextracted residues were not included when calculating half-lives to assess aquatic exposure in this assessment.

*Field Dissipation*

Several field dissipation studies were conducted in the United States and Canada. Field dissipation half-lives for thiamethoxam following broadcast applications ranged from 13-70.7 days. Field dissipation half-lives for thiamethoxam following applications to turf ranged from 1.05-78.8 days. A field dissipation half-life of 100 days was determined for an in-furrow application where the application rate was 4.2 times greater than the broadcast application rate. In California, Florida, and Michigan studies, quantifiable thiamethoxam residues were detected at a maximum depth of 6-12 inches following broadcast applications. In California, quantifiable thiamethoxam residues were detected at a 12-18 inches in turf plots and 18-24 inches in bare plots. In California and New Jersey studies, quantifiable residues were detected at 6-12 inches in turf plots and 12-18 inches in bare plots. In a study conducted at four test sites in Canada with thiamethoxam formulated as a seed treatment, thiamethoxam half-lives ranged from 72 to 111 days. It is important to note, the residue of concern, clothianidin (CGA-322704), discussed in **Section 2**, is also formed under field conditions.

Two aquatic field dissipation studies of thiamethoxam were conducted in Arkansas and Louisiana. These studies investigated the dissipation of thiamethoxam in a paddy water column (aquatic phase) and paddy soil (terrestrial phase) when thiamethoxam was applied as a seed treatment. In Arkansas, thiamethoxam dissipated in both phases with a calculated dissipation half-life of 11.6 days in paddy water and 26.7 days in paddy soil. No major degradates were detected in the paddy soil or water column. In Louisiana, thiamethoxam dissipated in both phases at a calculated dissipation half-life of 17.2 days in paddy water and 13.6 days in paddy soil. Major degradates CGA-355190 (10%) and CGA-353042 (10.2%) were observed in the water column. Field dissipation half-lives are similar to or within an order of magnitude of degradation half-lives conducted in the laboratory.

A summary of all available environmental fate data for thiamethoxam is provided in **Table 3-2**. Major degradate information including names, structures and percent formation identified in thiamethoxam laboratory and field studies are listed in **Table 3-3**.

Table 3-2. Environmental Fate Data for Thiamethoxam

| **Study** | **Value** | | **Major Degradates1,**  **Comments** | | **MRID(s)** |
| --- | --- | --- | --- | --- | --- |
| Hydrolysis  (t1/2; d) | 4.2 (pH 9)  8.4 (pH 9)  572 (pH 7; stable)  643 (pH 7; stable) | | NOA-404617 (35.2% AR; 21d; pH 9) CGA-355190 (60% AR; 30d; pH 9) | | 44703416  44703417 |
| Aqueous Photolysis  (t1/2; d) | 3.36 (pH 5)  3.90 (pH 5) | | CGA-353042 (67.3% AR; 30d.) | | 44715024  44715025 |
| Soil Photolysis  (t1/2; d) | 80  97 | | None | | 44715027  44715028 |
| Aerobic Soil Metabolism  (t1/2; d) | 294 (sandy loam; 25oC)  353 (sandy loam; 25oC)  101 (clay loam; 25oC)  60.1 (silt loam; 20oC)  174 (loamy sand; 20oC)  272 (loamy sand; 20oC)  188 (loamy sand; 20oC)  268 (sand; 20oC)  464 (sandy loam; 20oC)  110 (sandy loam; 20oC)  136 (loamy sand; 20oC)  73.6 (silt loam; 20oC)  143 (silt loam; 20oC)  34.3 (silt loam; 20oC) | | Clothianidin2 (19-37% AR)  CGA-355190 (23.7% AR; 365d.)  CO2 (21-44% AR) | | 44703419  44703501  44703418  49589503  49589504  49589505  49589506  49589506  49589506  49589506  49589506  49589507  49589507  49589507 |
| Anaerobic Soil Metabolism  (t1/2; d) | 81.3 (loam; 20oC)  76.2 (loam; 20oC)  77.7 (sandy clay loam; 20oC)  45.6 (sandy loam; 20oC)  118 (silt loam; 20oC) | | CGA-3227042 (17.3% AR; 30d.)  CGA-355190 (14-31% AR)  NOA-407475 (13.5-14.2% AR)  CO2 (14.2% AR; 120 d.) | | 49829901  49829902  49829902  49829902  49829902 |
| Anaerobic Aquatic Metabolism  (t1/2; d) | 28.6 (sandy loam-water; 25oC)  25.3 (sandy loam-water; 25oC)  20.7 (silt loam-water; 20oC) | | CGA-355190 (24-31% AR)  NOA-407475 (18-69% AR) | | 44715029  44715030  49589508 |
| Aerobic Aquatic Metabolism  (t1/2; d) | 16.2 (loam sediment-water; 25oC)  16.3 (loam sediment-water; 25oC)  35.1 (sandy loam-water; 25oC) | | CGA-355190 (79% AR; 115 d.)  NOA-407475 (22-52% AR)  NOA-404617 (36% AR; 21 d.)  CO2 (12-33% AR) | | 44715032  44715032  49589509 |
| **Study** | **Value** | | | | **MRID** |
| Batch Equilibrium | Soil | *KF* (L/kg) | *1/n* | *KFoc* (mL/goc) | 44703502 |
| sandy clay loam | 2.32 | 0.83 | 77.2 |
| loam | 0.90 | 0.82 | 53.1 |
| sandy loam | 0.71 | 0.84 | 176.7 |
| sand | 0.22 | 0.86 | 43.0 |
| loam | 0.65 | 0.80 | 38.3 |
| silty clay loam | 0.79 | 0.88 | 33.1 |
| Terrestrial Field Dissipation  Half-life  (DT50; d) | 72-111 (seed treatment)  13 (broadcast application)  70.7 (broadcast application)  100.4 (furrow application)  1.05 to 78.8 (turf) | | Residues contained within the 6-24” soil layers. CGA-3227042 forms under field conditions. | | 44703505  44727506  44948902  45086202  44948903 |
| Aquatic Field Dissipation  Half-life  (DT50; d) | 11.6 to 17.2 (paddy water)  13.6 to 26.7 (paddy soil) | | CGA-355190 (10%; LA)  CGA-353042 (10.2%; LA) | | 47558101  47558102  47558103 |

1 Major degradates are >10% of applied radioactivity in laboratory studies. 2 CGA-322704 = clothianidin

# Identification of Transformation Products of Concern

As discussed above, thiamethoxam may degrade into various transformation or degradation products through multiple pathways. One of thiamethoxam’s major degradation products is clothianidin (PC code 044309; also referred to as CGA-322704), which is also a registered neonicotinoid insecticide. Applications of clothianidin products are assessed independently in a separate biological evaluation. Of all the degradates of thiamethoxam, only clothianidin is considered to be of toxicological concern. This assessment considers exposures of thiamethoxam and clothianidin resulting from applications of thiamethoxam products.

In the majority of the available fate studies, clothianidin is formed as a minor degradate (<10% of the applied dose); however, it was identified as a major degradate (>10% of applied residue) in three of eight aerobic soil metabolism studies and one of two anaerobic soil metabolism studies (**Table 3-3**). Clothianidin is also formed under field conditions as it is detected in terrestrial field dissipation studies. In addition, clothianidin is formed as a plant metabolite and has been detected in leaf, pollen and nectar samples collected from thiamethoxam-treated crops (see USEPA 2020a for more details). Based on the available toxicity data (**Chapter 2**), clothianidin was included as a transformation product of concern in the aquatic modeling in addition to the parent, thiamethoxam. However, clothianidin’s contribution to the overall modeling half-lives was minimal. For example, the aquatic modeling half-live inputs increase from 236 to 383 days for aerobic soil metabolism and 29 to 32 days for anaerobic aquatic metabolism. Therefore, in the aquatic exposure analysis, thiamethoxam represents the majority of the residue. For terrestrial species, both thiamethoxam and clothianidin are considered residues of concern. Since clothianidin forms in plants, animals that consume plants may be exposed to both chemicals.

Several other compounds were also identified as major degradates in most of the available fate studies including: CGA-353042, CGA-335190, NOA-404617 and NOA-407475. When considering degradates of potential toxicological concern, it is assumed that the toxicity associated with thiamethoxam is attributed to the presence of the N-nitro group.[[1]](#footnote-2) Of the major degradates, only NOA-404617 maintains the N-nitro group, so, CGA-353042, CGA-335190, and NOA-407475 are assumed to be less toxic than the parent compound. Although the metabolite NOA-404617 contains the N-nitro group, it is not quantitatively assessed or considered a degradate of concern because it is not formed under conditions that are expected to contribute to exposures in natural aquatic habitats. This degradate was formed through hydrolysis under alkaline conditions (pH 9) (which is generally not representative of most aquatic environments and so is not accounted for in EFED’s standard aquatic modeling). NOA-404617 was also observed in one aerobic aquatic metabolism study; however, the modeling half-life of thiamethoxam is already on the order of months and so consideration of this degradate is not expected to impact the estimate of exposure from thiamethoxam and clothianidin[[2]](#footnote-3).; Therefore, NOA-404617 was not included as a degradate of toxicological concern.

Table 3-3. Thiamethoxam and Major Degradates A

| **Code Name (Synonym)** | **Chemical Name** | **Chemical Structure** | **Study Type** | **MRID** | **Maximum**  **%AR (day)** | **Final %AR**  **(study length)** |
| --- | --- | --- | --- | --- | --- | --- |
| **PARENT** | | | | | | |
| **Thiamethoxam**  **(CGA-293343)** | **CAS:** 3-[(2-chloro-5-thiazolyl)methyl]tetrahydro-5-methyl-N-nitro-4H-1,3,5-oxadiazin-4-imine  **CAS No.:** 153719-23-4  **Formula:** C8H10ClN5O3S  **MW:** 291.71 g/mol  **SMILES:** CN1COCN(C1=N[N+](=O)[O-])Cc2cnc(s2)Cl |  | PARENT | | | |
| **CGA-322704**  **(Clothianidin)** | **CAS:** Guanidine, N -[(2-chloro-5-thiazolyl)methyl]-N'-methyl-N''-nitro-  **CAS No.:** 131748-59-9  **Formula:** C6H8ClN5O2S  **MW:** 249.67 g/mol  **SMILES:** CN/C(=N/[N+](=O)[O-])/NCc1cnc(s1)Cl |  | Aerobic Soil Metabolism | 44703418 | 2.15% (90 d) | 1.37% (365 d) |
| 44703419 | 4.71% (365 d) | 4.71% (365 d) |
| 44703501 | 2.58% (182 d) | 1.61% (365 d) |
| 49589503 | **29.4%** (220 d) | **29.4%** (220 d) |
| 49589504 | 7.74% (120 d) | 7.74% (120 d) |
| 49589505 | 3.4% (118 d) | 3.4% (118 d) |
| 49589506 | **18.9% (121 d)** | **18.9% (121 d)** |
| 49589507 | **36.8% (90 d)** | **15.1% (363 d)** |
| Anaerobic Soil Metabolism | 49829901 | 7.2% (90 d) | 3.1% (153 d) |
| 49829902 | **17.3%** (30 d) | **10.1%** (120 d) |
| Anaerobic Aq. Metabolism | 44715029 | 1.1% (0 d) | ND (365 d) |
| 44715030  00 | 1.1% (0 d) | ND (365 d) |
| **CGA-353042** | **CAS:** 2H-1,3,5-Oxadiazine-4-amine, 3,6-dihydro-3-methyl |  | Aqueous Photolysis | 44715024 | **67.3% (30 d)** | **67.3% (30 d)** |
| **NOA-407475** | **CAS:** 4H-1,3,5-Oxadiazin-4-imine, 3-[(2-chloro-5-thiazolyl)methyl]tetrahydro-5-methyl-  **Formula:** C8H11ClN4OS  **MW:** 246.72 g/mol  **SMILES:** CN1COCN(C1=N)Cc2cnc(s2)Cl |  | Anaerobic soil metabolism | 49829901 | **14.2%** (153 d) | **14.2%** (153 d) |
| 49829902 | **13.5%** (120 d) | **13.5%** (120 d) |
| Aerobic Aquatic Metabolism | 44715032 | **52.0% (30 d)** | **29.8% (365 d)** |
| 49589509 | **21.8% (70 d)** | 6.99% (100 d) |
| Anaerobic Aquatic Metabolism | 44715031 | **69.1 (271 d)** | **63.0 (365 d)** |
| 49589508 | **17.6 (70 d)** | **15.6 (100 d)** |
| **CGA-355190** | **CAS:** 4H-1,3,5-Oxadiazin-4-one, 3-[(2-chloro-5-thiazolyl)methyl]tetrahydro-5-methyl-  **Formula:** C8H10ClN3O2S  **MW:** 247.7 g/mol  **SMILES:** CN1COCN(C1=O)Cc2cnc(s2)Cl |  | Hydrolysis | 44703417 | **59.5% (30 d)** | **59.5% (30 d)** |
| Aerobic Soil Metabolism | 44703418 | **23.7 (365 d)** | **23.7 (365 d)** |
| Anaerobic Soil Metabolism | 49829901 | **14.0%** (90 d) | 6.0% (153 d) |
| 49829902 | **31.0%** (120 d) | **31.0%** (120 d) |
| Aerobic Aquatic Metabolism | 44715032 | **78.9%** (115 d) | **46.6%** (365 d) |
| 49589509 | 6.92% (48 d) | 3.04% (100 d) |
| Anaerobic Aquatic Metabolism | 44715031 | **24.4 (180 d)** | **19.0 (365 d)** |
| 49589508 | **31.3 (48 d)** | **21.7 (100 d)** |
| **NOA-404617** | **CAS:** Urea, N-[(2-chloro-5-thiazolyl)methyl]-N'-nitro-  **Formula:** C5H5ClN4O3S  **MW:** 236.63 g/mol  **SMILES:** c1c(sc(n1)Cl)CNC(=O)N[N+](=O)[O-] |  | Hydrolysis | 44703416 | **35.2% (21 d)** | **33.3% (30 d)** |
| Anaerobic Soil Metabolism | 49829901 | 6.6% (120 d) | 0.8% (153 d) |
| 49829902 | 7.6% (120 d) | 7.6% (120 d) |
| Aerobic Aquatic Metabolism | 44715032 | **36.0%** (21 d) | 1.6% (365 d) |
| 49589509 | 8.00% (48 d) | 1.10% (100 d) |
| Anaerobic Aquatic Met. | 49589508 | 7.67% (48 d) | 2.47% (100 d) |
| 49829902 | 4.0% (30 d) | 0.5% (120 d) |
| Aerobic Soil Metabolism | 44703418 | 3.80 (365 d) | 3.80 (365 d) |
| Aerobic Aquatic Metabolism | 44715032 | 9.8% (365 d) | 9.8% (365 d) |

A Major = > 10% formation, AR means “applied radioactivity”. MW means “molecular weight”. ND = not detected

# Measures of Aquatic Exposure

In general, maximum application rates and minimum application retreatment intervals are modeled to estimate the exposure to thiamethoxam based on the use defined by the Preliminary Aquatic and Non-Pollinator Terrestrial Risk Assessment to Support Registration Review (EPA, 2017a) and addendum (EPA, 2020b), unless otherwise noted (**APPENDIX 1-2**). To streamline the assessment, use scenarios were grouped based on the relevant aquatic modeling scenario. Thiamethoxam-specific modeling scenarios are used for modeling each use (or crop group). 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 thiamethoxam uses included in the master use summary table (**APPENDIX 1-2**) by HUC 2 Regions (**Figure 3-1**) and by aquatic bin (2-7). 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)[[3]](#footnote-4) 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 1-2**.

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 (**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 some cases close to 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 thiamethoxam residues of concern, 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 3-4. 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.

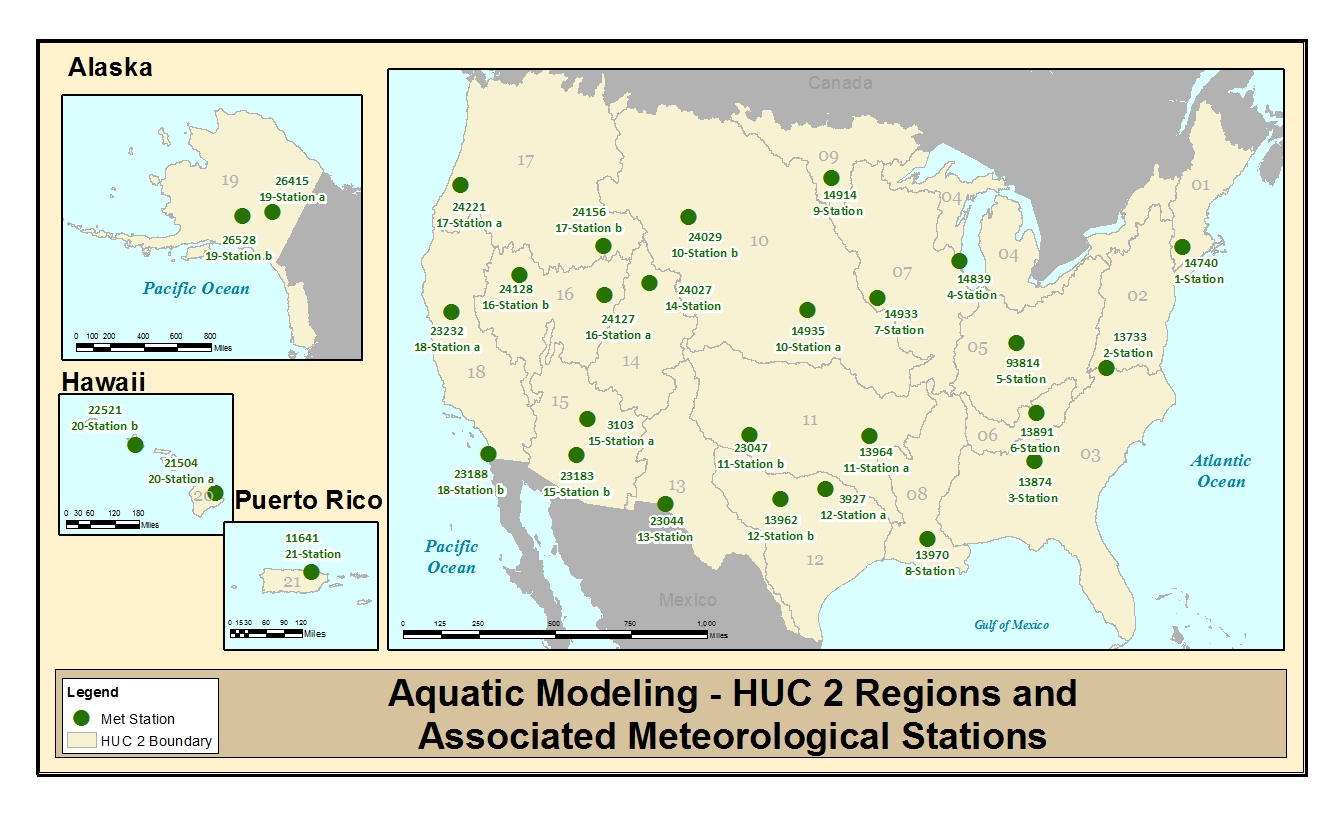
2 The habitat being evaluated is the reach or segment that abuts or is immediately adjacent to the treated field. This habitat is assumed to run the entire length of the treated area.

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).

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

## 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. 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 thiamethoxam 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 thiamethoxam applications (**APPENDIX 1-2**) including those that occur in both agricultural and non-agricultural settings.

Applications occur from aircraft, tractors and ATV’s for litter applications and application equipment include various sprayers (boom, airblast, hose-end, trigger, mist), spreaders, drenchers, injectors, irrigators (sprinkler and solid set), seed treaters, paintbrush and syringe (crack and crevice and spot treatments) and the traditional watering can.

Thiamethoxam is formulated as granule (G), water-dispersible granule (WDG), emulsifiable concentrate (EC), flowable concentrate (FC), ready to use (RTU), soluble concentrate/solid (SC/S), wettable powder (WP), agar gel (bait stations) and dust (D) for seed coatings/treatment.

### Spray Drift

The default spray drift inputs were assumed for all thiamethoxam applications. It should be noted that some registered labels (*e.g.* EPA Reg. No. 100-938) include spray drift precaution language that reads “*Do not cultivate or plant crops within 25 feet of the aquatic area as to allow growth of a vegetative buffer strip*.” Since this language is not on all registered labels, no spray drift adjustments were made for vegetative buffer strips. The default spray drift fractions used for all foliar spray in aquatic modeling are shown in **Table 3-5**. No spray drift is simulated for irrigation (chemigation) applications.

Table 3-5. 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** | **Default** | | |
| **Aerial** | **Airblast** | **Ground** |
| 1 | 10 | Wetland | 0.15 | 64 | 0.125 | 0.042 | 0.062 |
| 4 | 4 | Reservoir | 2.74 | 82 | 0.135 | 0.048 | 0.066 |
| 7 | 7 | Pond | 2 | 64 | 0.125 | 0.042 | 0.062 |
| 1parameters correspond to the input values used in PWC modeling.  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. | | | | | | | |

Some thiamethoxam labels specify the use of handheld application equipment (*e.g.*, hose-end sprayers, trigger sprayers, mist sprayers). Data are not available on the magnitude of spray drift that may result from these types of applications; however, these application methods are not expected to result in substantial drift. Generally, all crops that permit the use of such equipment also permit the use of ground boom or aerial equipment. Therefore, for purposes of quantitative exposure estimation, ground boom or aerial equipment, which tend to generate higher spray drift values, were chosen as conservative proxies for all application methods for the relevant crops.

### Application Timing

In selecting application dates for aquatic modeling, EPA considers several factors including label directions, timing of pest pressure, meteorological conditions, and pre-harvest restriction intervals. Agronomic information was consulted to determine the timing of crop emergence, pest pressure and seasons for different crops. General sources of information include crop profiles (<https://ipmdata.ipmcenters.org/#cropprofiles> ), agricultural extension bulletins, and/or available state-specific use information.

For most thiamethoxam uses, PWC model inputs for the application dates were based on the registered labels, the crop emergence and harvest timings specified in the PWC scenario, and precipitation data for the associated meteorological station. Application dates were selected to represent conservative and reasonable estimates.

Thiamethoxam directions for use indicate the timing of application is primarily dependent on pest pressure, directing the user to apply before pests reach damaging levels and retreat if pest populations rebuild to potentially damaging levels. Given this broad application timing language, the middle (*i.e.*, 15th day) of the month with the highest amount of precipitation (for the meteorological station for the PWC scenario) was chosen in order to generate a conservative estimate of exposure. Pesticide loading to surface water is directly affected by precipitation events. Once the first day of application was selected, minimum retreatment intervals were assumed to determine when subsequent applications would occur. Pre-harvest intervals (the minimum time between an application and harvest) were also considered. Applications would not occur closer to harvest than allowed by the pre-harvest interval. For details on application date selection for use of thiamethoxam, see **APPENDIX 1-3** and **APPENDIX 3-1**.

## Special Agricultural Considerations

### PFAM

Thiamethoxam is used on rice and cranberry and it is common practice to flood rice fields and cranberry bogs. Water from flooding a rice paddy or cranberry bog is generally released to an adjacent waterbody (wetland, cannel, river, stream, lake, or bay). The Pesticides in Flooded Applications Model (PFAM, version 2.0) was used to simulate rice and cranberry agricultural practices.

PFAM was developed specifically for regulatory applications to estimate exposure for pesticides used in flooded agriculture such as rice paddies and 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, allows for defining different receiving water bodies.

PFAM was used to estimate the concentration of thiamethoxam residues of concern in the flood water released from a rice paddy or cranberry bog. The reported concentrations represent water introduced to the field and not mixed with any additional water (*i.e.*, receiving water body). The concentration of thiamethoxam residues of concern 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 thiamethoxam residues of concern in the flood water to that in an adjacent waterbody depends on 1) the length of time thiamethoxam residues of concern are in the flooded paddy or bog, 2) the distance the water travels between the paddy or bog and the adjacent waterbody, 3) the amount of dilution and 4) whether the flood water is mixed with additional water that also contains thiamethoxam residues of concern. 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.

### Poultry Litter Applications

In addition to traditional agricultural applications, thiamethoxam can also be applied in poultry houses and livestock areas for fly and litter beetle control.

For poultry house use, the litter collected from the poultry house applications could potentially be used as a soil amendment after it has been treated with thiamethoxam. To assess the impacts of poultry litter use as a soil amendment, EPA modeled the amount of thiamethoxam 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 thiamethoxam was applied to the entire poultry house each time a treatment was 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 thiamethoxam treated manure on a corn field was developed using the approach used by Shamblen and Judkins (2012). An example calculation for three applications (per proposed product label; EPA Reg No. 70585-10) of Agita 10 WG partial (10%) house treatment of a 9000 ft2 poultry house (total treatment area = 3 app. x 900 ft2 = 2,700 ft2) follows:

1. Application rate for Agita 10 WG (Reg. No. 70585-10): 0.04 oz of Agita/ft2; treating a total of 2,700 ft2 = 108 oz Agita.
2. Agita contains 10% w/w thiamethoxam a.i.; 10% of 108 oz Agita = 10.8 oz = 0.675 lb thiamethoxam.
3. A typical poultry (broiler house) has six flocks of broilers before a full litter clean out, followed by storage, then application on a corn field. Six flocks will produce 168 tons of manure, and require 35 tons of bedding, resulting in a total of 203 tons of litter.
4. The cumulative residual concentration of thiamethoxam in litter is 0.675 lb a.i./203 tons litter = 0.0033 lb a.i./ton litter.

Since the thiamethoxam application rate can vary by area of poultry house treated and by the nitrogen requirement of the crop (mass applied as litter application), lower bound and upper bound scenarios were modeled for thiamethoxam residues of concern to characterize poultry litter application to agricultural fields.

The lower-bound application rate for thiamethoxam residues of concern-treated poultry litter for corn was estimated using the process described above except with a lower application rate of litter per acre. Poultry manure application rate information collected by the Biological and Economic Analysis Division (BEAD; USEPA 2017b) suggests that growers would rarely use more than 2-3 tons of litter per acre. This is due both to practicality (transportation costs, bulk handling, *etc*.) and the legal limits imposed by state nutrient management regulations, which are largely driven by phosphorous rather than nitrogen. (*i.e.*, phosphorous is limiting and precludes the high tonnage usage of manure for corn production). Therefore, based on a residual thiamethoxam concentration in litter of 0.0033 lbs. a.i./ton litter, and a typical manure application rate of 2.0 tons/A, the lower-bound outdoor equivalent application rate for thiamethoxam is 0.007 lbs. a.i./A (*i.e.*, 0.0033 lbs. a.i./ton \* 2.0 tons/A).

The upper-bound application rate for thiamethoxam residues of concern-treated poultry litter for corn was estimated using a high-end field application rate based on 8.0 tons/A litter treatment as a function of the nitrogen requirement for the crop and a (larger) 27,040 square feet treatment area. This application rate represents the maximum registered application rate for thiamethoxam (0.266 lbs. a.i./A) and approximates an order of magnitude greater treatment area than the lower-end field application rate (treating 2,700 sq. ft.)

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**) and scaled the application rate to reflect perimeter treatments.

### Seed Treatment

Thiamethoxam 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 thiamethoxam action area. As such, seed treatment applications were not assessed quantitatively, but are discussed qualitatively in **APPENDIX 4-5.** The exception for thiamethoxam is rice. As there is no foliar treatment permitted for rice, rice seed treatment was modeled, however because the poultry litter use layer encompasses applications to rice paddies, the EECs were summarized with those for poultry litter applications.

### Non-Agricultural Uses and Considerations

As described in thiamethoxam master use summary table (**APPENDIX 1-2**) non-agricultural use patterns include: turf (residential lawns, golf courses, sod fields, athletic fields, recreational areas), ornamentals (nurseries, field nurseries, non-bearing fruit and nut trees), Christmas tree plantations, residential and commercial perimeter treatment areas and bait stations. Thiamethoxam also has a number of 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.

#### 3.5.5.1 Exposure from Urban, Suburban and Homeowner Uses

Thiamethoxam 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 (livestock) outdoor buildings, nuisance pests – perimeter treatment, non-agricultural rights-of-way, 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 3-6. Application Information for Modeled Homeowner Scenario based on Maximum Labeled Application Rates

|  |  |
| --- | --- |
| **PWC Scenario** | **Residential** |
| Included uses | Garden, turf  House perimeter, along fences, patios, garbage cans, under porches, shrubbery, firewood piles, ornamentals, etc. |
| Application rate | 0.1374 lbs. a.i./A = 0.2661 x 0.88 (% total coverage area for perimeter uses minus driveway (2.8%) and house (9.2%) contributions (~12%) x 0.587 (watershed correction factor) |
| Application interval | Not applicable (single application) |
| **PWC Scenario** | **ROW** |
| Included uses | Commercial Perimeter, along landscapes, apartment buildings, correctional facilities, dumpsters, factories, fencing, greenhouses, railcars, restaurants, schools, warehouses, etc. |
| Application rate | 0.0066 lbs. a.i./A = 0.2661 x 0.042 (% total area for perimeter, pervious areas contributing for ROW scenario) x 0.587 (watershed correction factor) |
| Application interval | Not applicable (single application) |

1 EPA Reg. No. 100-1437: maximum application rate (0.266 lb. a.i./A)

EPA modeled residential uses employing the residential ESA scenario. EPA believes this approach is protective for all uses in a suburban/urban environment, as thiamethoxam is expected to be mainly applied to pervious surfaces in these environments and the scenarios account for runoff from both pervious and impervious surfaces. EPA employed this method based on comments received during the development of the Biological Evaluation for carbaryl.

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.  Summaries of the environmental fate model input parameters used in the PWC for the modeling of thiamethoxam is presented in **Table 3-7**. Input parameters are 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[[4]](#footnote-5) (USEPA, 2009),
* *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[5]](#footnote-6)* (NAFTA, 2012; USEPA, 2012c), and
* *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[6]](#footnote-7) (USEPA, 2013)
* *Methods for Assessing Aquatic Exposure to Residue(s) of Concern* (USEPA, 2019)

Table 3-7. Input Values Used for Tier II Surface Water Modeling with PWC or PFAM

| **Parameter (units)** | **Value** | **MRID** | **Comments** |
| --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (Koc (mL/g)) | 70.23 | 44703502 | average Koc of 6 values (77.2, 53.1, 176.7, 43.0, 38.3, 33.1) |
| Aerobic Aquatic Metabolism Half-life (days) 25°C | 34.4 | 44715032  49589509 | represents the 90th percentile confidence bound on the mean half-life of 3 values (16.2, 16.3, 35.1) |
| Anaerobic Aquatic Metabolism Half-life (days) 25°C | 32 | 44715029  44715030  49589508 | represents the 90th percentile confidence bound on the mean **total residue** half-life of 3 values (28.5, 27.4, 14.6b) |
| Aqueous Photolysis  Half-life @ pH 5 (days)  40o N latitude, 25oC | 4.46 | 44715024  44715025 | represents the 90th percentile confidence bound on the mean half-life of 2 values (3.36, 3.90) |
| Hydrolysis  Half-life (days), 25°C | 0 (stable) | 44703416  44703417 | estimated half-lives (572 and 643 days) are beyond the duration of the 30-d study, thus considered stable. |
| Aerobic Soil Metabolism  Half-life (days) 20°C | 383 a | 44703419  44703501  44703418  49589503  49589504  49589505  49589506  49589507 | represents the 90th percentile confidence bound on the mean **total residue** half-life of 14 values (802b, 649b, 180b, 166, 239, 330, 340, 401, 245, 138, 368, 165, 213, 127) |
| Molecular Weight (g/mol) | 291.7 | 44703304 | Product Chemistry |
| Vapor Pressure (torr or mm Hg) at 25oC | 4.95 x 10-11 | 44703305 | Product Chemistry |
| Solubility in Water  at 25oC, (mg/L) | 4,100 | 44703305 | Product Chemistry |
| Foliar Half-life (days) | 0 (stable) | -- | default value |
| Application Efficiency (fraction) | 0.95 (aerial)  0.99 (ground)  1.0 (chemigation) | -- | default values |
| Spray Drift (fraction) | varies by method and waterbody | AgDRIFT c | *See* Section 3.4.2 |
| Heat of Henry (J/mol) | 45,727 | EpiSuite d | 5,500 (from HenryWin for thiamethoxam) x 8.314 (constant) |
| a total residue half-life calculation includes the residue, clothianidin.  b Half-lives corrected using Q10 temperature correction equation.  c <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment#AgDrift>  d <https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface> | | | |

Application rates, methods, and timing for the different labeled uses are provided in **APPENDIX 3-1**. Application rates, scenarios, and timing for rice and cranberry applications are provided in **Table 3-7.**

## Aquatic Modeling Results

The estimated environmental concentrations (EECs) derived from the PWC modeling based on maximum labeled rates included in the master use summary table 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 rice and cranberry applications are summarized in **Table 3-10.** The complete set of modeling inputs and results are available in **APPENDIX 3-1**. PWC and PFAM 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 PWC Daily Average Water Column EECs for Thiamethoxam Residues of Concern

| **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 | 1.06 - 58 | 2 - 111 | 0.29 - 17.91 | 0.29 - 17.91 | 2 - 111 | 0.15 - 9.65 | 0.15 - 9.65 |
| HUC 2 | 0.57 - 171 | 2 - 108 | 0.18 - 17.29 | 0.18 - 17.29 | 2 - 108 | 0.09 - 6.62 | 0.09 - 6.62 |
| HUC 3 | 0.60 - 48 | 2 - 126 | 0.21 - 21.62 | 0.21 - 21.62 | 2 - 126 | 0.10 - 14.44 | 0.10 - 14.44 |
| HUC 4 | 1.00 - 61 | 2 - 101 | 0.22 - 8.49 | 0.22 - 8.49 | 2 - 101 | 0.10 - 4.48 | 0.10 - 4.48 |
| HUC 5 | 1.05 - 114 | 3 - 127 | 0.28 - 15.79 | 0.28 - 15.79 | 3 - 127 | 0.15 - 5.83 | 0.15 - 5.83 |
| HUC 6 | 0.62 - 46 | 2 - 143 | 0.14 - 16.67 | 0.14 - 16.67 | 2 - 143 | 0.07 - 7.42 | 0.07 - 7.42 |
| HUC 7 | 0.84 - 123 | 3 - 119 | 0.29 - 14.56 | 0.29 - 14.56 | 3 - 119 | 0.14 - 9.41 | 0.14 - 9.41 |
| HUC 8 | 0.93 - 197 | 4 - 151 | 0.55 - 21.49 | 0.55 - 21.49 | 4 - 151 | 0.32 - 16.68 | 0.32 - 16.68 |
| HUC 9 | 0.52 - 186 | 2 - 63 | 0.10 - 11.21 | 0.10 - 11.21 | 2 - 63 | 0.05 - 6.17 | 0.05 - 6.17 |
| HUC 10a | 1.44 - 223 | 3 - 126 | 0.47 - 17.78 | 0.47 - 17.78 | 3 - 126 | 0.25 - 9.57 | 0.25 - 9.57 |
| HUC 10b | 1.54 - 182 | 4 - 134 | 0.24 - 9.16 | 0.24 - 9.16 | 4 - 134 | 0.12 - 5.31 | 0.12 - 5.31 |
| HUC 11a | 1.24 - 72 | 3 - 145 | 0.45 - 24.44 | 0.45 - 24.44 | 3 - 145 | 0.26 - 15.69 | 0.26 - 15.69 |
| HUC 11b | 1.60 - 189 | 3 - 148 | 0.47 - 19.79 | 0.47 - 19.79 | 3 - 148 | 0.23 - 14.75 | 0.23 - 14.75 |
| HUC 12a | 1.07 - 91 | 3 - 145 | 0.20 - 22.14 | 0.20 - 22.14 | 3 - 145 | 0.10 - 12.44 | 0.10 - 12.44 |
| HUC 12b | 1.22 - 114 | 3 - 117 | 0.15 - 13.81 | 0.15 - 13.81 | 3 - 117 | 0.07 - 6.03 | 0.07 - 6.03 |
| HUC 13 | 1.12 - 203 | 4 - 147 | 0.07 - 7.73 | 0.07 - 7.73 | 4 - 147 | 0.03 - 4.76 | 0.03 - 4.76 |
| HUC 14 | 0.95 - 153 | 4 - 194 | 0.06 - 11.79 | 0.06 - 11.79 | 4 - 194 | 0.02 - 6.82 | 0.02 - 6.82 |
| HUC 15a | 1.26 - 122 | 5 - 234 | 0.32 - 14.86 | 0.32 - 14.86 | 5 - 234 | 0.15 - 7.99 | 0.15 - 7.99 |
| HUC 15b | 0.73 - 149 | 4 - 165 | 0.11 - 19.48 | 0.11 - 19.48 | 4 - 165 | 0.05 - 5.13 | 0.05 - 5.13 |
| HUC 16a | 0.72 - 141 | 5 - 213 | 0.09 - 8.51 | 0.09 - 8.51 | 5 - 213 | 0.04 - 4.51 | 0.04 - 4.51 |
| HUC 16b | 0.78 - 154 | 5 - 225 | 0.03 - 5.22 | 0.03 - 5.22 | 5 - 225 | 0.01 - 2.83 | 0.01 - 2.83 |
| HUC 17a | 0.91 - 35 | 3 - 115 | 0.24 - 13.56 | 0.24 - 13.56 | 3 - 115 | 0.10 - 12.52 | 0.10 - 12.52 |
| HUC 17b | 0.67 - 141 | 3 - 102 | 0.07 - 5.69 | 0.07 - 5.69 | 3 - 102 | 0.03 - 3.42 | 0.03 - 3.42 |
| HUC 18a | 0.44 - 64 | 4 - 173 | 0.10 - 11.08 | 0.10 - 11.08 | 4 - 173 | 0.04 - 6.85 | 0.04 - 6.85 |
| HUC 18b | 0.67 - 129 | 4 - 144 | 0.08 - 7.82 | 0.08 - 7.82 | 4 - 144 | 0.03 - 4.62 | 0.03 - 4.62 |
| HUC 19a | 1.21 - 166 | 3 - 110 | 0.11 - 7.92 | 0.11 - 7.92 | 3 - 110 | 0.05 - 4.32 | 0.05 - 4.32 |
| HUC 19b | 0.59 - 162 | 3 - 108 | 0.14 - 16.99 | 0.14 - 16.99 | 3 - 108 | 0.07 - 7.16 | 0.07 - 7.16 |
| HUC 20a | 1.48 - 56 | 4 - 168 | 0.53 - 22.26 | 0.53 - 22.26 | 4 - 168 | 0.26 - 13.89 | 0.26 - 13.89 |
| HUC 20b | 1.21 - 177 | 3 - 125 | 0.25 - 17.21 | 0.25 - 17.21 | 3 - 125 | 0.14 - 6.42 | 0.14 - 6.42 |
| HUC 21 | 1.38 - 84 | 4 - 162 | 0.57 - 33.45 | 0.57 - 33.45 | 4 - 162 | 0.30 - 23.48 | 0.30 - 23.48 |

Table 3-9. Range of PWC Pore Water EECs for Thiamethoxam Residues of Concern

| **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 | 2 - 112 | 0.12 - 12.85 | 0.12 - 12.85 | 2 - 112 | 0.07 - 4.52 | 0.07 - 4.52 |
| HUC 2 | 2 - 108 | 0.06 - 26.55 | 0.06 - 26.55 | 2 - 108 | 0.03 - 4.22 | 0.03 - 4.22 |
| HUC 3 | 2 - 126 | 0.05 - 89.45 | 0.05 - 89.45 | 2 - 126 | 0.03 - 14.98 | 0.03 - 14.98 |
| HUC 4 | 2 - 101 | 0.06 - 22.49 | 0.06 - 22.49 | 2 - 101 | 0.03 - 3.54 | 0.03 - 3.54 |
| HUC 5 | 3 - 131 | 0.08 - 16.37 | 0.08 - 16.37 | 3 - 131 | 0.05 - 2.76 | 0.05 - 2.76 |
| HUC 6 | 2 - 144 | 0.04 - 19.18 | 0.04 - 19.18 | 2 - 144 | 0.01 - 4.99 | 0.01 - 4.99 |
| HUC 7 | 3 - 119 | 0.05 - 21.72 | 0.05 - 21.72 | 3 - 119 | 0.03 - 4.93 | 0.03 - 4.93 |
| HUC 8 | 4 - 151 | 0.08 - 229.76 | 0.08 - 230 | 4 - 151 | 0.06 - 54.88 | 0.06 - 54.88 |
| HUC 9 | 2 - 66 | 0.03 - 19.36 | 0.03 - 19.36 | 2 - 66 | 0.02 - 2.96 | 0.02 - 2.96 |
| HUC 10a | 3 - 127 | 0.08 - 14.95 | 0.08 - 14.95 | 3 - 127 | 0.06 - 3.17 | 0.06 - 3.17 |
| HUC 10b | 4 - 135 | 0.07 - 7.59 | 0.07 - 7.59 | 4 - 135 | 0.04 - 1.74 | 0.04 - 1.74 |
| HUC 11a | 4 - 145 | 0.06 - 36.86 | 0.06 - 36.86 | 4 - 145 | 0.05 - 5.65 | 0.05 - 5.65 |
| HUC 11b | 3 - 148 | 0.08 - 8.44 | 0.08 - 8.44 | 3 - 148 | 0.06 - 2.39 | 0.06 - 2.39 |
| HUC 12a | 3 - 145 | 0.06 - 17.61 | 0.06 - 17.61 | 3 - 145 | 0.03 - 3.23 | 0.03 - 3.23 |
| HUC 12b | 3 - 118 | 0.04 - 37.86 | 0.04 - 37.86 | 3 - 118 | 0.02 - 6.72 | 0.02 - 6.72 |
| HUC 13 | 4 - 148 | 0.01 - 4.33 | 0.01 - 4.33 | 4 - 148 | 0.01 - 1.23 | 0.01 - 1.23 |
| HUC 14 | 4 - 195 | 0.03 - 3.15 | 0.03 - 3.15 | 4 - 195 | 0.01 - 1.83 | 0.01 - 1.83 |
| HUC 15a | 5 - 236 | 0.07 - 13.08 | 0.07 - 13.08 | 5 - 236 | 0.05 - 3.72 | 0.05 - 3.72 |
| HUC 15b | 4 - 167 | 0.02 - 4.32 | 0.02 - 4.32 | 4 - 167 | 0.01 - 1.78 | 0.01 - 1.78 |
| HUC 16a | 5 - 215 | 0.05 - 3.02 | 0.05 - 3.02 | 5 - 215 | 0.02 - 1.67 | 0.02 - 1.67 |
| HUC 16b | 5 - 227 | 0.01 - 2.47 | 0.01 - 2.47 | 5 - 227 | 0.00 - 1.42 | 0.00 - 1.42 |
| HUC 17a | 3 - 118 | 0.07 - 11.61 | 0.07 - 11.61 | 3 - 118 | 0.03 - 5.21 | 0.03 - 5.21 |
| HUC 17b | 3 - 103 | 0.02 - 1.72 | 0.02 - 1.72 | 3 - 103 | 0.01 - 1.11 | 0.01 - 1.11 |
| HUC 18a | 4 - 175 | 0.03 - 17.15 | 0.03 - 17.15 | 4 - 175 | 0.01 - 4.45 | 0.01 - 4.45 |
| HUC 18b | 4 - 145 | 0.02 - 4.15 | 0.02 - 4.15 | 4 - 145 | 0.01 - 1.37 | 0.01 - 1.37 |
| HUC 19a | 3 - 111 | 0.03 - 2.43 | 0.03 - 2.43 | 3 - 111 | 0.02 - 1.63 | 0.02 - 1.63 |
| HUC 19 b | 3 - 108 | 0.05 - 3.77 | 0.05 - 3.77 | 3 - 108 | 0.03 - 2.98 | 0.03 - 2.98 |
| HUC 20a | 4 - 168 | 0.06 - 7.83 | 0.06 - 7.83 | 4 - 168 | 0.05 - 3.26 | 0.05 - 3.26 |
| HUC 20b | 3 - 125 | 0.04 - 1.77 | 0.04 - 1.77 | 3 - 125 | 0.03 - 1.40 | 0.03 - 1.40 |
| HUC 21 | 4 - 162 | 0.07 - 14.23 | 0.07 - 14.23 | 4 - 162 | 0.05 - 4.22 | 0.05 - 4.22 |

Table 3-10. PFAM EECs for Rice and Cranberry

| **PFAM Scenario**  **Application Date**  **Application Rate (lbs. a.i./A)** | **HUC 2** | **State** | **Daily Average EECs (µg/L)** |
| --- | --- | --- | --- |
| **Rice** | | | |
| ECO MS Winter.PFS  05/02 (0.17) | 3, 6 | Mississippi | 15.7 |
| ECO MS noWinter.PFS  05/02 (0.17) | 15.8 |
| ECO MO Winter.PFS  05/05 (0.17) | 5, 7 | Missouri | 23.3 |
| ECO MO noWinter.PFS  05/05 (0.17) | 23.5 |
| ECO AR Winter.PFS  05/01 (0.17) | 8 | Arkansas | 21.8 |
| ECO AR noWinter.PFS  05/01 (0.17) | 21.9 |
| ECO LA Winter.PFS  04/10 (0.17) | Louisiana | 23.0 |
| ECO LA noWinter.PFS  04/10 (0.17) | 23.0 |
| ECO MO Winter.PFS  05/05 (0.17) | Missouri | 23.3 |
| ECO MO noWinter.PFS  05/05 (0.17) | 23.5 |
| ECO MS Winter.PFS  05/02 (0.17) | Mississippi | 15.7 |
| ECO MS noWinter.PFS  05/02 (0.17) | 15.8 |
| ECO AR Winter.PFS  05/01 (0.17) | 11 | Arkansas | 21.8 |
| ECO AR noWinter.PFS  05/01 (0.17) | 21.9 |
| ECO MO Winter.PFS  05/05 (0.17) | Missouri | 23.3 |
| ECO MO noWinter.PFS  05/05 (0.17) | 23.5 |
| ECO TX Winter.PFS  04/09 (0.17) | Texas | 22.9 |
| ECO TX noWinter.PFS  04/09 (0.17) | 22.9 |
| ECO LA Winter.PFS  04/10 (0.17) | 12 | Louisiana | 23.0 |
| ECO LA noWinter.PFS  04/10 (0.17) | 23.0 |
| ECO TX Winter.PFS  04/09 (0.17) | Texas | 22.9 |
| ECO TX noWinter.PFS  04/09 (0.17) | 22.9 |
| ECO CA Winter.PFS  05/02 (0.17) | 16, 17, 18 | California | 25.0 |
| **Cranberry** | | | |
| MA Cranberry Winter Flood.PFS  11/15 (0.063), 11/22 (0.063), 11/29 (0.063) | 1 | Massachusetts  Maine  Connecticut  Rhode Island  New Hampshire  Vermont | 35.0 |
| MA Cranberry Winter Flood.PFS  11/15 (0.188) | 30.0 |
| MA Cranberry Winter Flood.PFS  11/15 (0.063), 11/22 (0.063), 11/29 (0.063) | 2 | New Jersey  New York  Delaware  Pennsylvania  Vermont | 35.0 |
| MA Cranberry Winter Flood.PFS  11/15 (0.188) | 30.0 |
| WI Cranberry Winter Flood.PFS  06/15 (0.063), 06/22 (0.063), 06/29 (0.063) | 4 | Wisconsin  Michigan  New York | 21.1 |
| WI Cranberry Winter Flood.PFS  06/15 (0.188) | 20.5 |
| WI Cranberry Winter Flood.PFS  06/15 (0.063), 06/22 (0.063), 06/29 (0.063) | 5 | New York  Pennsylvania | 21.1 |
| WI Cranberry Winter Flood.PFS  06/15 (0.188) | 20.5 |
| WI Cranberry Winter Flood.PFS  06/15 (0.063), 06/22 (0.063), 06/29 (0.063) | 7 | Wisconsin  Missouri | 21.1 |
| WI Cranberry Winter Flood.PFS  06/15 (0.188) | 20.5 |
| WI Cranberry Winter Flood.PFS  06/15 (0.063), 06/22 (0.063), 06/29 (0.063) | 9 | Minnesota | 21.1 |
| WI Cranberry Winter Flood.PFS  06/15 (0.188) | 20.5 |
| OR Cranberry No Flood.PFS  12/15 (0.063), 12/22 (0.063), 12/29 (0.063) | 17a  17b | Washington  Oregon | 0.997 |
| OR Cranberry No Flood.PFS  12/15 (0.188) | 0.937 |
| OR Cranberry Winter Flood.PFS  12/15 (0.063), 12/22 (0.063), 12/29 (0.063) | 56.8 |
| OR Cranberry Winter Flood.PFS  12/15 (0.188) | 67.3 |
| OR Cranberry No Flood.PFS  12/15 (0.063), 12/22 (0.063), 12/29 (0.063) | 18a  18b | Oregon | 0.997 |
| OR Cranberry No Flood.PFS  12/15 (0.188) | 0.937 |
| OR Cranberry Winter Flood.PFS  12/15 (0.063), 12/22 (0.063), 12/29 (0.063) | 56.8 |
| OR Cranberry Winter Flood.PFS  12/15 (0.188) | 67.3 |

## Available Monitoring Data

Examination of the EPA 303(d) list of impaired waters[[7]](#footnote-8) on May 10, 2021, indicates no impairments caused by thiamethoxam.

Water monitoring data were obtained from the National Water Quality Monitoring Council’s Water Quality Data Portal [[8]](#footnote-9) (USEPA and USGS), 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 thiamethoxam use occurs. Also, the sampling sites, as well as the number of samples, vary by year. The vulnerability of the sampling site to thiamethoxam 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 thiamethoxam use; as such, peak concentrations of thiamethoxam 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 thiamethoxam 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 thiamethoxam.

### Water Quality Portal

Comprehensive surface water and groundwater thiamethoxam data were obtained in March 2021 in a download of data from the Water Quality Data Portal (<http://www.waterqualitydata.us/>). **Table 3-11** provides a summary of the results by HUC 2 region, with sampling occurring from 2009 to 2021 at over 1,690 sites in most HUC 2 regions.

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

| **HUC-21** | **Years** | **Number of Sites** | **Number of Samples** | **Number of Samples Labeled Non-Detections** | **Measured Detection Range (µg/L)**2 |
| --- | --- | --- | --- | --- | --- |
| 01 | -- | -- | -- | -- | -- |
| 02 | 2012 - 2019 | 141 | 338 | 313 | 3.40E-03 - 2.80E-01 |
| 03 | 2011 - 2021 | 499 | 1604 | 1426 | 8.20E-03 - 2.70E+00 |
| 04 | 2010 - 2018 | 89 | 317 | 286 | 3.40E-03 - 1.26E-01 |
| 05 | 2014 | 1 | 1 | 1 | NA |
| 06 | -- | -- | -- | -- | -- |
| 07 | 2010 - 2019 | 381 | 2334 | 2146 | 3.40E-03 - 1.92E+00 |
| 08 | -- | -- | -- | -- | -- |
| 09 | 2010 - 2019 | 106 | 403 | 375 | 3.90E-03 - 2.14E-01 |
| 10 | 2009 - 2020 | 86 | 625 | 578 | 3.40E-03 - 1.98E-01 |
| 11 | 2012 - 2019 | 7 | 186 | 186 | NA |
| 12 | 2012 | 1 | 1 | 1 | NA |
| 13 | 2009 - 2019 | 12 | 67 | 67 | NA |
| 14 | 2009 - 2020 | 49 | 230 | 228 | 2.41E-02 - 2.98E-02 |
| 15 | 2010 - 2019 | 55 | 171 | 171 | NA |
| 16 | 2010 - 2020 | 4 | 29 | 29 | NA |
| 17 | 2009 - 2018 | 20 | 31 | 30 | 2.09E-02 - 2.09E-02 |
| 18 | 2011 - 2020 | 237 | 993 | 837 | 2.77E-03 - 5.00E+00 |
| 19 | -- | -- | -- | -- | -- |
| 20 | -- | -- | -- | -- | -- |
| 21 | 2013 | 2 | 2 | 2 | NA |

1 Note that historical monitoring data does not reflect updates to labels or commitment letters. 2 NA = no detections. – no samples recorded

### 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 are higher but mostly within an order of magnitude of values reported from the Water Quality Portal. Maximum values more than an order of magnitude higher than comparable Water Quality Portal values are sampled from direct runoff (*i.e.,* ponded water on a treated field or wetland directly adjacent to treated field). Hladik *et al.* (2014) sampled in the U.S. midwest and found a maximum thiamethoxam concentration of 0.19 µg/L. Miles *et al.* 2017 sampled in Tippecanoe Co., Indiana and found a maximum concentration of 0.02 µg/L. Struger *et al.* (2017) sampled in southern Ontario, Canada and found a maximum concentration of 1.34 µg/L. Main *et al.* 2013sampled in a Saskatchewan wetland and found a maximum concentration of1.49 µg/L. Morrissey *et al.* 2015 conducted an international review of many monitoring studies and found a maximum concentration of 63.4 µg/L in ponded water on a treated field in Quebec.

## Aquatic Exposure Summary

Model-derived EECs represent an upper bound on potential exposure as a result of the use of thiamethoxam residues of concern. 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 flow rate 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.

Some registered labels (*i.e.* EPA Reg. No. 100-938) include spray drift precaution language that reads “*Do not cultivate or plant crops within 25 feet of the aquatic area as to allow growth of a vegetative buffer strip*.” If the “do not *plant*” restriction for a vegetative buffer strip is considered similar to the “do not *apply”* assumption made when applying a spray drift buffer, aquatic exposure from spray drift should also be reduced. For example, applying a 25 ft buffer to the default spray drift values for the standard farm pond reduces deposition from 12.5 to 9.2% for aerial applications and 6.2 to 2.7% for ground applications, respectively. Consequently, aquatic EECs would also be reduced by no more than 3.3% for aerial applications and 3.5% for ground applications. While vegetative filter strips are directionally correct and associated spray drift reductions can be characterized, there is uncertainty in quantifying the reductions resulting from runoff, as soil organic matter content, water holding capacity, slope, and vegetation can all impact the ability of a filter strip to retain pesticide.

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.

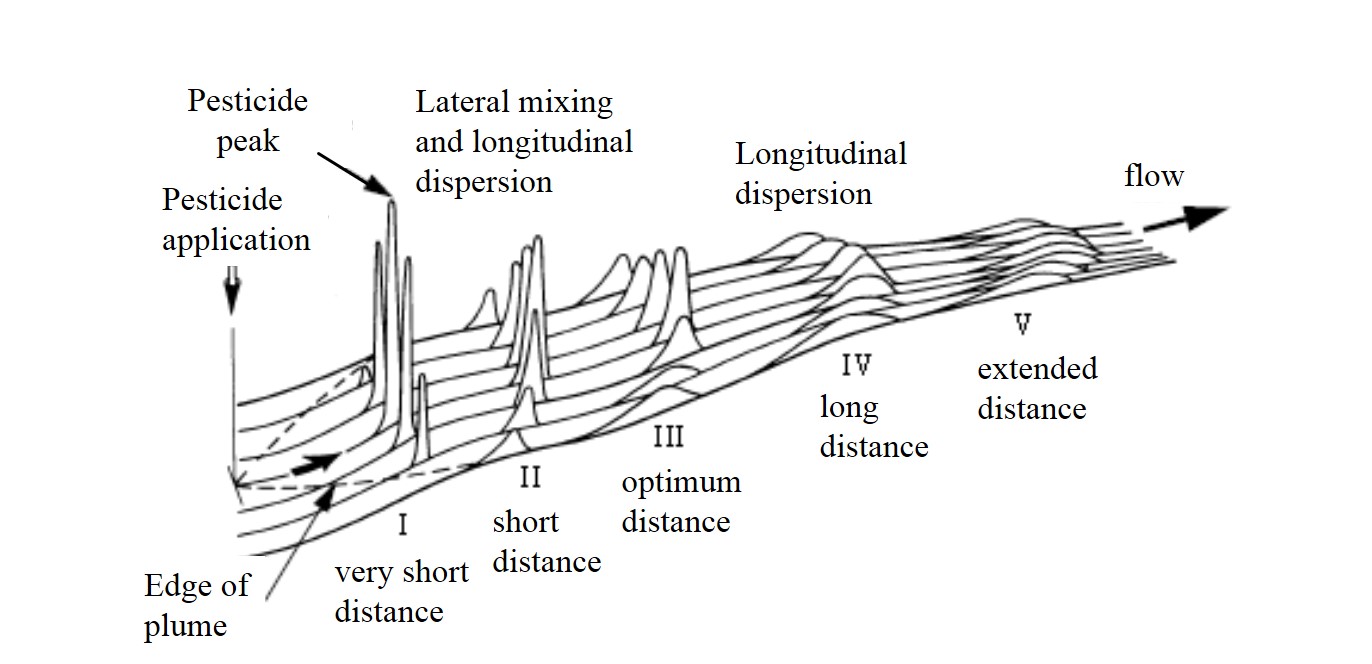


Figure 3-2. Effect of Pesticide Concentration via Advective Dispersion

## 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 thiamethoxam residues of concern 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 thiamethoxam’s log Kow value (-0.13), 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. In this assessment, a default foliar dissipation half-life of 35 days is used for terrestrial modeling purposes since suitable foliar dissipation data specific to thiamethoxam are not available.

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 thiamethoxam (a lower bound application rate of 0.05 lb. a./A with 1 application per year and an upper bound application rate of 0.266 lb. a.i./A with 1 application per year) and are provided below in **Table 3-12**. 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), could alter the EECs used to assess a species exposure. All uses for thiamethoxam and associated application rates are provided in **APPENDIX 1-2**. **Table 3-12** summarizes the mean and upper bound dietary-based EECs and the associated base model that is used in the MAGtool to predict the EECs. Thiamethoxam uses also include granular formulations and seed treatments; 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**  **(0.05 lb. a.i./A x 1 applications/year)** | | **Upper bound application rate**  **0.266 lb. a.i./A x 1 application/year)** | |
| **Upper Bound** | **Mean** | **Upper Bound** | **Mean** |
| Short Grass | T-REX | 12 | 4.3 | 63.8 | 22.6 |
| Tall Grass, nectar and pollen | T-REX | 5.5 | 2 | 29.3 | 9.6 |
| Broadleaf plants | T-REX | 7.0 | 2.3 | 35.9 | 12.0 |
| Seeds, fruit and pods | T-REX | 1.0 | 0.35 | 4 | 1.9 |
| Arthropods (above ground) | T-REX | 5.0 | 3.3 | 25 | 17.3 |
| Soil-dwelling invertebrates (earthworms) | Earthworm fugacity | 0.00079 | NA | 0.0042 | NA |
| Small mammals (15 g, short grass diet) | T-HERPS | 11.9 | 4.2 | 63.3 | 22.4 |
| Large mammals (1000 g, short grass diet)2 | T-HERPS | 1.9 | 0.68 | 10.1 | 3.6 |
| Small birds (20 g, insect diet) | T-HERPS | 21.4 | 14.8 | 113.6 | 78.6 |
| Small terrestrial phase amphibians or reptiles (2 g; insect diet) | T-HERPS | 1.0 | 0.72 | 5.5 | 3.8 |
| 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 thiamethoxam in aquatic dietary items

# Literature Cited

For Master Record Identification (MRID) Number citations refer to **APPENDIX 2-4** OPPIN bibliography.

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1. The Metabolism Assessment Review Committee (MARC) suggested the toxicity associated with thiamethoxam was due to the presence of the N-nitro group (USEPA, 1999). [↑](#footnote-ref-2)
2. Previous modeling efforts indicated little influence on EECs when the aerobic aquatic metabolism half-life was assumed to be stable. [↑](#footnote-ref-3)
3. The exposure models can be found at: <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-4)
4. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling> (accessed May 2021) [↑](#footnote-ref-5)
5. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values> (accessed May 2021) [↑](#footnote-ref-6)
6. The draft guidance is available at www.regulations.gov docket number: EPA-HQ-OPP-2013-0676 [↑](#footnote-ref-7)
7. <https://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T> [↑](#footnote-ref-8)
8. <https://www.waterqualitydata.us/> [↑](#footnote-ref-9)