Chapter 3 – Clothianidin Exposure Characterization

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# Environmental Transport and Fate Characterization

Clothianidin has a high solubility, low octanol-water partitioning coefficient, low vapor pressure, and low Henry’s Constant (**Table 3-1**). These data suggest that clothianidin has a low potential for volatilization and bioaccumulation. The main routes of dissipation are soil photolysis, aqueous photolysis, and runoff. The major routes transporting clothianidin from treatment sites to off-site habitats include runoff and spray drift. The preponderance of clothianidin surface water detections are in agricultural areas and in the vicinity of local use areas.

Table 3-1. Physical and Chemical Properties of Clothianidin

|  |  |
| --- | --- |
| **Property** | **Value** |
| Molecular Weight (g/mol) | 249.7 |
| Water Solubility@25°C mg/L | 327 |
| Vapor Pressure@ 25°C Torr | 1.3x10-15 |
| Henrys Law Constant (atm-m3 mole-1) | 2.9x10-16 |
| log Kow (unitless) | 1.12 |
| Dissociation Constants | Data shows no dissociation from pH 2-12 |

*Degradation and Metabolism*

The major route of dissipation for clothianidin appears to be photolysis, with an aqueous photolysis half-life less than 1 day (MRID 45422318) and a soil photolysis half-life of 34 days (MRID 45422323). Major photolysis degradates include: TZMU, MU, MG, MIT, FA, and HMIO. A 33-day study (MRID 45422317) concluded that clothianidin was stable to hydrolysis at all tested pHs and temperatures. A subsequent study (MRID 47483003) found degradation at elevated temperatures, with estimated extrapolated half-lives at 40°C and 50°C of 1,733 and 1,155 days, respectively. Clothianidin is persistent in soil (Goring *et al*, 1975), with aerobic half-lives ranging from 144 to 5,357 days (ten soils, MRIDs 45422325 and 45422326). The only major degradate observed was MNG (maximum of 10.7%). Clothianidin is considered persistent in aerobic aquatic environments, with aerobic half-lives of 178 and 180 days (two water/sediment systems, MRID 46826903). The major degradate was TMG (maximum of 24.5%). Clothianidin degrades in anaerobic aquatic environments, with a half-life of 27 days (MRID 45422330). The major degradate was unextracted residues (maximum of 83%).

*Soil sorption and mobility*

Clothianidin is considered mobile to moderately mobile (FAO, 2000). Soil organic carbon partition coefficient (Koc) values ranged from 84 to 129 L/kgoc for all laboratory test soils except for a sandy loam soil, which had a Koc value of 345 L/kgoc (MRID 45422311). A time-dependent sorption study was conducted on sandy loam and silt loam soils (MRID 45422312), evaluating the mobility of clothianidin for up to 99 days. Sorption appeared to increase over time, as Koc values increased from 205 (low dose) and 153 (high dose) L/kgoc at Day 0 to 582 (low dose) and 323 (high dose) L/kgoc at Day 99 in the sandy loam soil. In the silt loam soil, Koc values increased from 120 (low dose) and 98 (high dose) L/kgoc at Day 0 to 413 (low dose) and 311 (high dose) L/kgoc at Day 99. It should be noted that at the end of the study clothianidin comprised 56.3% and 58.0% of the applied radioactivity in the sandy loam and silt loam soils, respectively. Degradates were not identified.

*Field dissipation*

Clothianidin is expected to dissipate very slowly under terrestrial field conditions, based on the results of five bare ground field experiments conducted in the United States and Canada. Half-lives of clothianidin, based on residues in the 0-15 cm soil depth, were 277 days (Wisconsin sand soil, incorporated), 315 days (Ohio silt loam soil, not incorporated), 365 days (Ontario silt loam soil, incorporated), and 1,386 days (North Dakota clay loam soil, not incorporated), and could not be determined at a fifth site due to limited dissipation during the 25-month study (Saskatchewan silty clay loam soil, incorporated). Soil incorporation of the clothianidin application did not appear to be a significant factor in determining the rate of dissipation. Clothianidin was generally not detected below the 45 cm soil depth except at one site, where it moved into the 45-60 cm depth. No degradates were detected at >10% of the applied, and degradates were generally only detected in the 0-15 cm soil layer. However, in many cases most of the parent compound remained untransformed at the close of the study; further accumulation of degradates could have occurred. The substantial amount of clothianidin remaining in the soil profile at the termination of these studies indicates that further leaching of clothianidin may occur in following years if sufficient precipitation occurs. Two aquatic field dissipation studies were conducted for clothianidin, one in California and one in Louisiana (MRID 48364804, MRID 48364803). The maximum concentration at Day 0, pre-irrigation, was 210 μg/L in the Louisiana plot cropped with rice. Clothianidin residues quickly dissipated to an average of 74.1 μg/L three days after application, and further dissipated to 0.7 μg/L, 28 days after application. The California plots show similar dissipation results with a maximum clothianidin concentration of 51.9 μg/L at Day 0, quickly dissipating to an average of 1.1 μg/L ten days after application in the plots cropped with rice.

The majority of major degradates were formed in the aqueous photolysis study (MRID 45422318), where six major degradates formed: MG at 35%, TZMU at 40%, HMIO at 27%, MU at 11%, FA at 40%, and MIT at 16%. Of these, only TZMU formed as a major degradate in another study, 10% in the terrestrial field dissipation study (MRID 45422333), while it formed as a minor degradate in both the aerobic soil metabolism and aerobic aquatic metabolism studies (MRIDs 45422325, 45422326 and 46826903). MNG formed at 11% in the aerobic soil metabolism study (MRID 45422325), while TMG formed at a maximum of 25% in the aerobic aquatic metabolism study (MRID 46826903). Unextracted residues formed at a maximum of 83% in the anaerobic aquatic environments (MRID 45422330), but were not formed at levels greater than 10% in the aerobic metabolism studies.

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

Table 3-2. Environmental Fate Data for Clothianidin

| **Study** | **Value** | **Major Degradates**1**,****Comments** | **MRID #** |
| --- | --- | --- | --- |
| Abiotic HydrolysisHalf-life | Stable | None | 45422317  |
| Direct Aqueous Photolysis | 0.6 d | MG 34.7% (432 hours) TZMU, 29.3- 39.7% % (24 hours) HMIO 26.6% (24 hours) MU, 11.0% (432 hours) FA 39.7% (24 hours), MIT 16.1% (120 hours)  | 45422318 45422320 45422322  |
| Soil Photolysis Half-life | 34 d | None  | 45422323  |
| Aerobic Soil MetabolismHalf-life (20°C) | 233 d (Laacher silt loam, 0.86% OC) 144 d (Hofchen silt, 2.66% OC) 1670 d (Hanhofen loamy sand, 2.5% OC) 4029 d (Indiana sandy loam, 1.12% OC) 4196 d (Crosby silt loam, 1.37% OC) 1254 d (Elder loam, 1.41% OC) 5357 d (Fuguay loamy sand, 0.35% OC) 545 d (Quincy loamy sand, 0.4% OC) 551 d (Sparta sand, 0.73% OC) 716 d (Susan silt loam, 3.27% OC)  | NTG, 6.7% (Day 120) TZNG, 9.1% (Day 120) TZMU, ≤2.4% (Day 120)  | 45422325 45422325 45422325 45422325 45422326 45422326 45422326 45422326 45422326 45422326  |
| Anaerobic Aquatic MetabolismHalf-life (20°C) | 27 d (Silt loam, 6.2% OC)  | Unextracted residues, 83% (Day 360)  | 45422330  |
| Aerobic Aquatic MetabolismHalf-life (20°C) | 178 d (Loam, 0.9% OC) 182 d (Loam, 0.9% OC)  | TMG, 24.5% (Day 91), 13.8% at (Day 120)  | 46826903  |
| **Study** | **Value** | **MRID #** |
| Batch Equilibrium | *Soil* | *KF* (L/kg) | *1/n* | *KFoc* (mL/goc) | 45422311  |
| Loamy sand | 0.52 | 0.835 | 129 |
| sandy loam | 4.14 | 0.809 | 345 |
| Clay loam | 1.48 | 0.822 | 123 |
| Sandy loam | 1.77 | 0.815 | 84 |
| sand | 0.59 | 0.865 | 119 |
| Sandy loam |  |  |  | 45422312 – desorption only |
| Sandy loam | 0.258 | 0.901 | 25.3 | 45422313 |
| sand | 0.0199 | 0.702 | 5.2 |
| Silt loam | 0.193 | 0.925 | 16.5 |
| Sandy loam | 0.374 | 0.908 | 34.3 |
| sand | 0.0514 | 1.10 | 21.4 |
| Sandy loam | 5.36 | 0.849 | 525 | 45422316\* degradate TMG |
|  | sand | 2.44 | 0.826 | 642 |  |
|  | Silt loam | 15.8 | 0.780 | 1350 |  |
|  | Sandy loam | 39.5 | 0.730 | 3620 |  |
|  | sand | 14.8 | 0.780 | 6160 |  |
| **Study** | **Value** | **MRID #** |
| Terrestrial Field DissipationHalf-life | 277 d | Wisconsin sand | 45422332  |
| 1386 d | North Dakota clay loam | 45422334  |
| 315 d | Ohio silt loam | 45422333  |
| 365 d | Ontario silt loam | 45422335  |
| Not quantified due to variation | California sandy loam | 45490703  |
| 257 d | Washington loamy sand | 45490704  |
| 990 d | Georgia loamy sand | 45490705  |

1 Major degradates are defined as those which reach >10% of the applied radioactivity.

# Identification of Transformation Products of Concern

Available toxicity data for aquatic taxa indicate that, in general, the degradates were similar to (non-toxic) or less toxic than parent clothianidin. However, TMG is of concern to benthic invertebrates. Because the mobility of clothianidin and its degradates indicate that they do not readily bind to soil or sediment, unextracted residues were not considered for further analysis. Therefore, the stressors of concern for the aquatic assessment are determined to be parent clothianidin as well as the degradate TMG. TMG forms via aerobic aquatic metabolism and occurs at up to 25% of applied clothianidin. Consideration of the potential increased toxicity of formulations is considered through the selection of toxicity endpoints and is discussed further in **Chapter 2**.

# Measures of Aquatic Exposure

In general, maximum application rates and minimum application retreatment intervals are modeled to estimate the exposure to clothianidin based on the use defined by the Preliminary Aquatic and Non-Pollinator Terrestrial Risk Assessment to Support Registration Review (EPA, 2017) and addendum (EPA, 2020) unless otherwise noted (**APPENDIX 1-2**). To streamline the assessment, use scenarios were grouped based on the relevant aquatic modeling scenario.

Clothianidin-specific modeling simulations 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 clothianidin uses included in the Clothianidin Preliminary Risk Assessment (PRA) (**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 models used in this assessment are 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 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 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 clothianidin, 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 3-3** provides a crosswalk of the bins and how they were modeled.

Table 3-3. 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.

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. 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, use on citrus is only permitted in Florida so only HUC-03 was modelled for that use pattern.

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 clothianidin 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 clothianidin applications included in the Clothianidin PRA (**APPENDIX 1-2**) including those that occur in both agricultural and non-agricultural settings. Clothianidin may be applied to crops via a variety of methods including aerial and ground foliar sprays, soil treatment (*e.g.*, drench), chemigation (*e.g.*, soil incorporation or foliar), and as a seed treatment. Clothianidin is used on a wide array of agricultural crops, including: root and tuber vegetables, leafy vegetables, brassica, cucurbits, fruiting vegetables, cereal grains, citrus fruit, pome fruit, stone fruit, berries, tree nuts, beans and other legumes, herbs, oilseed crops (*e.g.,* canola, cotton), and tobacco. Additionally, there are non-agricultural uses including application to turf, poultry houses, and ornamentals. Clothianidin has 46 active Section 3 registered labels with various physical forms and co-formulated active ingredients.

Clothianidin 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 post-harvest applications, all applications are simulated considering the appropriate application timing (*e.g.*, dormant, foliar, and post-harvest applications to a crop) and label directions. However, application during the wettest month is assumed unless it is not reasonable.

### Spray Drift

The default spray drift inputs were assumed for all clothianidin applications as buffers are not required on labels. The default spray drift fractions used for all foliar spray in modeling are shown in **Table 3-4**. Spray drift based on a ground application is assumed for non-agricultural applications.

Table 3-4. 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** | **Ground** | **Airblast** |
| 4 | 4 | Reservoir | 2.74 | 82 | 0.135 | 0.066 | 0.048 |
| 7 | 7 | Pond | 2 | 64 | 0.125 | 0.062 | 0.042 |
| 1 | 10 | Wetland | 0.15 | 64 | 0.125 | 0.062 | 0.042 |
| 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 clothianidin labels specify the use of handheld application equipment (*e.g.*, hose-end sprayers, hand bulb dusters). 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](https://gcc02.safelinks.protection.outlook.com/?url=https%3A%2F%2Fipmdata.ipmcenters.org%2F%23cropprofiles&data=04%7C01%7CShelby.Andrew%40epa.gov%7Ce46f06f48150459d585808d918694ae2%7C88b378b367484867acf976aacbeca6a7%7C0%7C0%7C637567662805943349%7CUnknown%7CTWFpbGZsb3d8eyJWIjoiMC4wLjAwMDAiLCJQIjoiV2luMzIiLCJBTiI6Ik1haWwiLCJXVCI6Mn0%3D%7C1000&sdata=B6lQoRQ8xR9krrjhDwGxqancPqSrikVi69D%2BOfNqBk4%3D&reserved=0)), agricultural extension bulletins, and/or available state-specific use information.

Clothianidin may be applied during different seasons, and the directions for use indicate the timing of application is often associated with pest pressure*.* For clothianidin uses, PWC model inputs for the application dates were chosen based on precipitation data for the associated meteorological station. Application dates were selected to represent conservative and reasonable estimates. When choosing an application date within a time window (*i.e*., crop emergence or foliar application), the 15th of the month with the highest amount of precipitation (for the meteorological station for the PWC scenario) for that time window was chosen. 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. If multiple types of applications were allowed on one crop within one year, such as pre-plant or soil incorporation along with a foliar application(s), the retreatment interval was selected to reflect the specified timings. 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 clothianidin, 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 planted in rotation (*e.g.*, wheat), or that may be grown in multiple crop seasons per year. In addition, fallow applications are permitted. However, since the total use of clothianidin is restricted on an annual basis these agricultural practices should not result in more than 0.4 lb of clothianidin per acre per year for agricultural uses.

### PFAM

Clothianidin is used on rice and it is common practice to flood rice fields. Water from flooding rice patties 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 rice 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 clothianidin in the flood water released from a rice patty. The reported concentrations represent water introduced to the field and not mixed with any additional water (*i.e.*, receiving water body). The concentration of clothianidin 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 clothianidin in the flood water to that in an adjacent waterbody depends on 1) the length of time clothianidin is in the flooded patty, 2) the distance the water travels between the patty and the adjacent waterbody, 3) the amount of dilution and 4) whether the flood water is mixed with additional water that also contains clothianidin.

### 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, clothianidin 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 clothianidin. To assess the impacts of poultry litter use as a soil amendment, EPA modeled the amount of clothianidin 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 clothianidin-treated manure on a corn field was developed using the following process (Shamblen and Judkins, 2012):

1. Application rate for Darlex (Reg. No. 1021-2771) - 4 oz of Darlex/1000 ft2; treating a 20,000 ft2 house (maximum size poultry house) = 80 oz Darlex.
2. Darlex contains 23.6% w/w clothianidin a.i. 80 oz Darlex = 1.33125 lb clothianidin.
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 7.9875 lb of clothianidin (6 x 1.33125 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 clothianidin in litter is 7.9875 lb/203 tons litter = 0.039347 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 7920 lb of nitrogen.
8. Only 90% of the nitrogen is available to plants in the first year (USDA estimate of mineralization), resulting in 7128 lb of plant available nitrogen.
9. An additional 50% of the nitrogen is lost during application, resulting in 3564 lb plant-available nitrogen.
10. Based on the nitrogen application rate of 220 lb N/A, this results in 16.2 A being needed for the manure from six flocks (3564 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 clothianidin concentration in litter of 0.039347 lb a.i./ton litter, and a litter application rate of 12.5 tons/A, the outdoor equivalent application rate for clothianidin is 0.49 lb a.i./A.

Alternative poultry house treatment scenarios were suggested by the registrant (MRID 49681202) and OPP’s Biological and Economic Analysis Division (BEAD, USEPA, 2017) for modeling and are also considered in this biological evaluation.

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

Clothianidin 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, other grains (barley, wheat, *etc*.), and guayule seed. In the case of corn and other grains, because the application of treated poultry litter is assessed to corn and other grain fields (see previous section), EPA believes the EECs from this modeling are protective of corn and other grain seed treatment. In the case of guayule, these plants are primarily grown in Pinal County, Arizona and could occur in the same areas as where other crops can be grown and are covered in this assessment. As such, the seed treatment only uses do not impact the overlap extent of the clothianidin action area and were not modeled, but are discussed qualitatively in **APPENDIX 4-5**.

### Non-Agricultural Uses and Considerations

Clothianidin 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, airports, ornamental and/or shade trees, residential lawns, gardens, residential fruit and nut trees, golf courses, along fences, porches *etc*. When considered together, uses such as these could encompass a substantial fraction of an urban or suburban watershed. **Table 3-5** summarizes these uses, rates, intervals for different areas. Clothianidin also has a number of indoor uses. As EFED does not have usage 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.

Table 3-5. Application information for modeled homeowner scenario based on maximum labeled application rates

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **PWC Scenario** | **Developed** | **Developed Open Space** | **Golf** | **Orchards** | **OtherCrop** | **Residential** |
| Included uses | Commercial/Industrial Storage and Premise, Residential fruit tree, Airports | Ornamentals (except trees) | Golf Course | Fruit and Nut Trees (Residential) | Sod Farms | Residential (Outdoor) |
| Application rate (lb a.i./A) | 0.4, 1x | 0.4, 1x | 0.4, 1x | 0.4, 1x | 0.4, 1x | 0.4, 1x |

EPA modeled residential uses employing the Residential ESA scenarios. EPA believes this approach is protective for all uses in a suburban/urban environment, as clothianidin 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, EECs for residential uses were adjusted by a factor of 0.587. EECs were further adjusted for applications that are not expected to apply to an entire residential lot. Applications at commercial sites are further adjusted by 0.19 and applications to ornamentals are further adjusted by 0.694 consistent with their use in the 2016 pyrethroid/pyrethrin assessment (EPA, 2016e). Applications to residential trees are further adjusted by 0.11 consistent with the residential ESA scenario as used in the 2015 Biological Evaluation for malathion (EPA, 2016c).

## 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 clothianidin is presented in **Table 3-6**. 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[[2]](#footnote-3) (USEPA, 2009),
* *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[3]](#footnote-4)* (NAFTA, 2012; USEPA, 2012c), and
* *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[4]](#footnote-5) (USEPA, 2013)

Table 3-6. Input Values Used for Tier II Surface Water Modeling with PWC or PFAM

| **Parameter (units)** | **Value (s)** | **Source** | **Comments** |
| --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (Koc (L/kg o.c.)) | 161  | MRIDs 45422311-16  | Average of five values. The coefficient of variation for Koc values was lower than Kd values.  |
| Water Column Metabolism Half-life or Aerobic Aquatic Metabolism Half-life (days) 25°C | 554  | MRID 46826903  | The 90 percent upper confidence limit on the mean of two aerobic aquatic metabolism half-life values including clothianidin and TMG residues. |
| Benthic Metabolism Half-life or Anaerobic Aquatic Metabolism Half-life (days) 25°C | 81 8 | MRID 45422330  | Single value (27 days) times 3  |
| Aqueous Photolysis Half-life @ pH 7 (days)  | 0.6  | MRID 45422322  | Phoenix, AZ summer sunlight  |
| Hydrolysis Half-life (days) | Stable | MRID 45422317  |  |
| Soil Half-life or Aerobic Soil Metabolism Half-life (days) 25°C | 2709  | MRIDs 45422325, 45422326  | The 90 percent upper confidence limit on the mean of ten aerobic soil metabolism half-life values.  |
| MWT or Molecular Weight (g/mol) | 249.7  | 45422301  |  |
| Vapor Pressure (Torr) at 25oC | 1.3E-15 | 45422301  |  |
| Solubility in Water @ 25oC, pH not reported (mg/L) | 327  | 45422301  |  |
| Foliar Half-life (days) | 0 | default |  |
| Application Efficiency | 0.99 (ground)0.95 (air) | default |  |
| Drift | Default | AgDRIFT | See section 3.4.2 |
| Heat of Henry (J/mol) | 179,000 | Calculated | Boiling point = 412.5oC |

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 3-6**.

## 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 3-7** and **Table 3-8**, for water column and pore water, respectively. EECs for rice applications are summarized in **Table 3-9**. 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 3-7. Range of PWC Daily Average Water Column EECs for Toxic Residues of Clothianidin

| **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 | 1.9 - 28.5 | 3 - 104 | 0.56 - 22.19 | 0.56 - 22.19 | 3 – 104 | 0.36 - 20.58 | 0.36 - 20.58 |
| HUC 2 | 1.16 - 89.7 | 6 - 93 | 0.69 - 11.23 | 0.69 - 11.23 | 6 – 93 | 0.31 - 7.18 | 0.31 - 7.18 |
| HUC 3 | 1.46 - 47.6 | 4 - 152 | 0.59 - 24.82 | 0.59 - 24.82 | 4 – 152 | 0.32 - 16.55 | 0.32 - 16.55 |
| HUC 4 | 2.01 - 26.1 | 4 - 85 | 0.59 - 13.49 | 0.59 - 13.49 | 4 – 85 | 0.30 - 6.90 | 0.30 - 6.90 |
| HUC 5 | 2.04 - 62 | 6 - 144 | 0.69 - 18.42 | 0.69 - 18.42 | 6 – 144 | 0.43 - 10.82 | 0.43 - 10.82 |
| HUC 6 | 1.35 - 37.4 | 4 - 168 | 0.36 - 15.16 | 0.36 - 15.16 | 4 – 168 | 0.18 - 9.42 | 0.18 - 9.42 |
| HUC 7 | 1.66 - 68.6 | 4 - 96 | 0.54 - 20.34 | 0.54 - 20.34 | 4 – 96 | 0.31 - 14.47 | 0.31 - 14.47 |
| HUC 8 | 1.8 - 44.1 | 5 - 174 | 0.94 - 27.71 | 0.94 - 27.71 | 5 – 174 | 0.57 - 23.61 | 0.57 - 23.61 |
| HUC 9 | 1.1 - 109 | 2 - 65 | 0.36 - 9.526 | 0.36 - 9.526 | 2 – 65 | 0.19 - 5.70 | 0.19 - 5.70 |
| HUC 10a | 2.89 - 125 | 5 - 132 | 0.83 - 26.06 | 0.83 - 26.06 | 5 – 132 | 0.55 - 14.58 | 0.55 - 14.58 |
| HUC 10b | 2.18 - 107 | 5 - 137 | 0.59 - 14.67 | 0.59 - 14.67 | 5 – 137 | 0.33 - 7.57 | 0.33 - 7.57 |
| HUC 11a | 2.74 - 41.2 | 5 - 135 | 1.03 - 26.89 | 1.03 - 26.89 | 5 – 135 | 0.63 - 18.04 | 0.63 - 18.04 |
| HUC 11b | 3.1 - 57 | 5 - 133 | 0.90 - 24.7 | 0.90 - 24.7 | 5 – 133 | 0.49 - 12.71 | 0.49 - 12.71 |
| HUC 12a | 2.15 - 58.9 | 7 - 150 | 0.70 - 21.02 | 0.70 - 21.02 | 7 – 150 | 0.33 - 12.29 | 0.33 - 12.29 |
| HUC 12b | 2.75 - 40.7 | 6 - 141 | 0.50 - 14.9 | 0.50 - 14.9 | 6 – 141 | 0.25 - 10.91 | 0.25 - 10.91 |
| HUC 13 | 2.18 - 84 | 7 - 205 | 0.33 - 10.16 | 0.33 - 10.16 | 7 – 205 | 0.15 - 5.77 | 0.15 - 5.77 |
| HUC 14 | 2.56 - 58.1 | 7 - 243 | 0.29 - 12.29 | 0.29 - 12.29 | 7 – 243 | 0.13 - 6.38 | 0.13 - 6.38 |
| HUC 15a | 2.77 - 54.7 | 8 - 275 | 0.38 - 18.46 | 0.38 - 18.46 | 8 – 275 | 0.25 - 10 | 0.25 - 10 |
| HUC 15b | 2.54 - 81.8 | 8 - 220 | 0.36 - 10.16 | 0.36 - 10.16 | 8 – 220 | 0.19 - 5.49 | 0.19 - 5.49 |
| HUC 16a | 1.72 - 101 | 7 - 260 | 0.17 - 10.75 | 0.17 - 10.75 | 7 – 260 | 0.13 - 5.62 | 0.13 - 5.62 |
| HUC 16b | 1.73 - 93.7 | 5 - 270 | 0.11 - 5.55 | 0.11 - 5.55 | 5 – 270 | 0.04 - 2.99 | 0.04 - 2.99 |
| HUC 17a | 1.76 - 35.7 | 5 - 117 | 0.49 - 15.6 | 0.49 - 15.6 | 5 – 117 | 0.22 - 12.9 | 0.22 - 12.9 |
| HUC 17b | 1.37 - 86.4 | 4 - 110 | 0.13 - 4.65 | 0.13 - 4.65 | 4 – 110 | 0.08 - 2.45 | 0.08 - 2.45 |
| HUC 18a | 1.27 - 35.9 | 8 - 217 | 0.39 - 15.46 | 0.39 - 15.46 | 8 – 217 | 0.25 - 8.95 | 0.25 - 8.95 |
| HUC 18b | 1.52 - 76.9 | 8 - 200 | 0.25 - 8.624 | 0.25 - 8.624 | 8 – 200 | 0.12 - 4.87 | 0.12 - 4.87 |
| HUC 19a | 2.02 - 54.7 | 4 - 102 | 0.29 - 10.65 | 0.29 - 10.65 | 4 - 102 | 0.19 - 6.34 | 0.19 - 6.34 |
| HUC 19b | 1.41 - 41.7 | 4 - 100 | 0.43 - 12.59 | 0.43 - 12.59 | 4 - 100 | 0.23 - 9.44 | 0.23 - 9.44 |
| HUC 20a | 3.06 - 44.4 | 5 - 167 | 1.17 - 26.82 | 1.17 - 26.82 | 5 - 167 | 0.60 - 17.62 | 0.60 - 17.62 |
| HUC 20b | 2.64 - 48.8 | 5 - 142 | 0.68 - 21.53 | 0.68 - 21.53 | 5 - 142 | 0.53 - 20.72 | 0.53 - 20.72 |
| HUC 21 | 3.08 - 27.5 | 6 - 164 | 1.28 - 45.97 | 1.28 - 45.97 | 6 - 164 | 0.73 - 36.99 | 0.73 - 36.99 |

Table 3-8. Range of PWC Pore Water EECs for Toxic Residues of Clothianidin

| **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 - 105 | 0.34 - 19.44 | 0.34 - 19.44 | 4 - 105 | 0.22 - 8.13 | 0.22 - 8.13 |
| HUC 2 | 6 - 96 | 0.33 - 18.85 | 0.33 - 18.85 | 6 - 96 | 0.17 - 4.37 | 0.17 - 4.37 |
| HUC 3 | 4 - 153 | 0.2 - 48.12 | 0.2 - 48.12 | 4 - 153 | 0.12 - 9.53 | 0.12 - 9.53 |
| HUC 4 | 4 - 97 | 0.22 - 16.73 | 0.22 - 16.73 | 4 - 97 | 0.12 - 3.42 | 0.12 - 3.42 |
| HUC 5 | 7 - 162 | 0.27 - 21.1 | 0.27 - 21.1 | 7 - 162 | 0.15 - 5.58 | 0.15 - 5.58 |
| HUC 6 | 4 - 169 | 0.14 - 20.51 | 0.14 - 20.51 | 4 - 169 | 0.07 - 4.67 | 0.07 - 4.67 |
| HUC 7 | 5 - 121 | 0.28 - 32.12 | 0.28 - 32.12 | 5 - 121 | 0.20 - 8.91 | 0.20 - 8.91 |
| HUC 8 | 5 - 177 | 0.27 - 328.5 | 0.27 - 328.5 | 5 - 177 | 0.18 - 82.79 | 0.18 - 82.79 |
| HUC 9 | 3 - 73 | 0.14 - 14.95 | 0.14 - 14.95 | 3 - 73 | 0.08 - 3.01 | 0.08 - 3.01 |
| HUC 10a | 5 - 135 | 0.32 - 11.53 | 0.32 - 11.53 | 5 - 135 | 0.19 - 5.60 | 0.19 - 5.60 |
| HUC 10b | 5 - 140 | 0.24 - 5.97 | 0.24 - 5.97 | 5 - 140 | 0.12 - 2.74 | 0.12 - 2.74 |
| HUC 11a | 5 - 141 | 0.27 - 22.8 | 0.27 - 22.8 | 5 - 141 | 0.19 - 5.43 | 0.19 - 5.43 |
| HUC 11b | 5 - 137 | 0.25 - 9.23 | 0.25 - 9.23 | 5 - 137 | 0.14 - 4.18 | 0.14 - 4.18 |
| HUC 12a | 7 - 152 | 0.20 - 14.76 | 0.20 - 14.76 | 7 - 152 | 0.12 - 3.62 | 0.12 - 3.62 |
| HUC 12b | 6 - 143 | 0.18 - 28.98 | 0.18 - 28.98 | 6 - 143 | 0.08 - 6.57 | 0.08 - 6.57 |
| HUC 13 | 7 - 208 | 0.10 - 5.54 | 0.10 - 5.54 | 7 - 208 | 0.04 - 2.93 | 0.04 - 2.93 |
| HUC 14 | 7 - 249 | 0.15 - 4.52 | 0.15 - 4.52 | 7 - 249 | 0.06 - 2.40 | 0.06 - 2.40 |
| HUC 15a | 8 - 283 | 0.20 - 14.66 | 0.20 - 14.66 | 8 - 283 | 0.11 - 4.25 | 0.11 - 4.25 |
| HUC 15b | 8 - 224 | 0.12 - 4.40 | 0.12 - 4.40 | 8 - 224 | 0.06 - 1.99 | 0.06 - 1.99 |
| HUC 16a | 7 - 266 | 0.07 - 4.48 | 0.07 - 4.48 | 7 - 266 | 0.05 - 2.30 | 0.05 - 2.30 |
| HUC 16b | 5 - 276 | 0.05 - 3.45 | 0.05 - 3.45 | 5 - 276 | 0.02 - 1.93 | 0.02 - 1.93 |
| HUC 17a | 5 - 126 | 0.19 - 18.2 | 0.19 - 18.2 | 5 - 126 | 0.08 - 6.76 | 0.08 - 6.76 |
| HUC 17b | 4 - 115 | 0.05 - 2.07 | 0.05 - 2.07 | 4 - 115 | 0.03 - 0.96 | 0.03 - 0.96 |
| HUC 18a | 8 - 219 | 0.14 - 4.57 | 0.14 - 4.57 | 8 - 219 | 0.09 - 3.34 | 0.09 - 3.34 |
| HUC 18b | 8 - 202 | 0.09 - 3.03 | 0.09 - 3.03 | 8 - 202 | 0.04 - 1.74 | 0.04 - 1.74 |
| HUC 19a | 5 - 104 | 0.14 - 5.11 | 0.14 - 5.11 | 5 - 104 | 0.08 - 3.09 | 0.08 - 3.09 |
| HUC 19 b | 4 - 103 | 0.17 - 7.03 | 0.17 - 7.03 | 4 - 103 | 0.12 - 5.00 | 0.12 - 5.00 |
| HUC 20a | 5 - 168 | 0.23 - 10.91 | 0.23 - 10.91 | 5 - 168 | 0.20 - 5.89 | 0.20 - 5.89 |
| HUC 20b | 5 - 143 | 0.17 - 5.19 | 0.17 - 5.19 | 5 - 143 | 0.13 - 6.32 | 0.13 - 6.32 |
| HUC 21 | 7 - 165 | 0.38 - 8.59 | 0.38 - 8.59 | 7 - 165 | 0.34 - 10.72 | 0.34 - 10.72 |

Table 3-9. PFAM EECs for Toxic Residues of Clothianidin

|  |  |  |  |
| --- | --- | --- | --- |
| **Scenario****Application Date** **(Rate lb a.i./A)** | **HUC 2** | **State** | **Daily Average EECs (µg/L)** |
| ***Rice, Foliar Post-flood Application (0.075 lbs a.i./A x 1)*** ***Application date selected to allow for the maximum retention of pesticide in paddy water***  |
|  |  |  |  |
| ECO CA Winter5/18 | 16, 17, 18 | California | 82.7 |
| ECO AR noWinter5/3 | 11, 08 | Arkansas | 82.7 |
| ECO LA dry seeded5/13  | 08, 12 | Louisiana | 82.7 |
| ECO MO noWinter7/5  | 07, 05, 08, 11 | Missouri | 82.7 |
| ECO MS noWinter (winter)5/30  | 03, 06, 08 | Mississippi | 55.1 |
| ECO TX Winter4/15  | 11, 12 | Texas | 82.7 |

## Available Monitoring Data

Examination of the EPA 303(d) list of impaired waters[[5]](#footnote-6) on May 17, 2021, indicates no impairments caused by clothianidin.

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 clothianidin use occurs. Also, the sampling sites, as well as the number of samples, vary by year. The vulnerability of the sampling site to clothianidin 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 clothianidin use; as such, peak concentrations of clothianidin 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 clothianidin 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 clothianidin.

### Water Quality Portal

Comprehensive surface water and groundwater clothianidin data were obtained in April 2021 in a download of data from the Water Quality Data Portal (<http://www.waterqualitydata.us/>). **Table 3-10** provides a summary of the results by HUC 2 region, with sampling occurring from 2011 to 2021 at over 1,400 sites in most HUC 2 regions. Monitoring data for degradate TMG are not currently available through the Water Quality Data Portal.

Table 3-10. Water Quality Portal Monitoring Data Summarized by 2-digit HUC for Clothianidin

| **HUC-2** | **Years** | **Number of Sites** | **Number of Samples** | **Number of Samples Labeled Non-Detections** | **Measured Detection Range (µg/L)** |
| --- | --- | --- | --- | --- | --- |
| 01 | -- | -- | -- | -- | -- |
| 02 | 2012 - 2019 | 141 | 338 | 314 | 3.00E-03 - 6.20E-02 |
| 03 | 2011 - 2021 | 509 | 1677 | 1443 | 2.00E-03 - 3.90E-01 |
| 04 | 2012 - 2018 | 86 | 313 | 257 | 3.90E-03 - 3.33E-01 |
| 05 | 2014 | 1 | 1 | 1 | 6.20E-031 |
| 06 | -- | -- | -- | -- | -- |
| 07 | 2011 - 2019 | 268 | 1968 | 1634 | 3.90E-03 - 2.60E-01 |
| 08 | -- | -- | -- | -- | -- |
| 09 | 2011 - 2019 | 83 | 350 | 329 | 6.20E-03 - 1.41E-01 |
| 10 | 2012 - 2020 | 60 | 425 | 375 | 3.90E-03 - 1.19E-01 |
| 11 | 2012 - 2019 | 7 | 180 | 180 | 6.20E-031 |
| 12 | 2012 | 1 | 1 | 1 | 6.20E-031 |
| 13 | 2013 - 2019 | 6 | 42 | 42 | 2.00E-021 |
| 14 | 2013 - 2020 | 33 | 140 | 140 | 3.90E-031 |
| 15 | 2013 - 2019 | 37 | 127 | 127 | 2.00E-021 |
| 16 | 2013 - 2020 | 3 | 14 | 14 | 3.90E-031 |
| 17 | 2013 - 2018 | 16 | 28 | 24 | 8.77E-05 - 1.13E-02 |
| 18 | 2011 - 2020 | 202 | 895 | 835 | 4.00E-03 - 1.34E+00 |
| 19 | -- | -- | -- | -- | -- |
| 20 | -- | -- | -- | -- | -- |
| 21 | 2013 | 2 | 2 | 2 | 6.20E-031 |

1 No detections, minimum detection limit reported

### Open Literature 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 via Water Quality Portal. Maximum values more than an order of magnitude higher than comparable Water Quality Portal values are sampled from direct run off (*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 concentration of 0.257 µg/L. Miles *et al* 2017 sampled in Tippecanoe Co., Indiana and found a maximum concentration of 0.67 µg/L. Struger *et al* (2017) sampled in southern Ontario, Canada and found a maximum concentration of 0.399 µg/L. Main *et. al.* 2013sampled in a Saskatchewan wetland and found a maximum concentration of3.11 µg/L. Morrissey *et. al.* 2015 conducted an international review of many monitoring studies and found a maximum concentration of 55.7 µg/L in ponded water on a treated field in Quebec. As discussed previously in section one, two aquatic field dissipation studies were conducted for clothianidin, one in California and one in Louisiana (MRID 48364804, MRID 48364803). While these studies are not open literature, they exhibit the highest monitored concentrations of clothianidin due to its direct application to water. The maximum concentration at Day 0, pre-irrigation, was 210 μg/L in the Louisiana plot cropped with rice. Clothianidin residues quickly dissipated to an average of 74.1 μg/L three days after application, and further dissipated to 0.7 μg/L, 28 days after application. The California plots show similar dissipation results with a maximum clothianidin concentration of 51.9 μg/L at Day 0, quickly dissipating to an average of 1.1 μg/L ten days after application in the plots cropped with rice.

## Aquatic Exposure Summary

Model-derived EECs represent an upper bound on potential exposure as a result of the use of clothianidin. 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

### 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 in 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 clothianidin 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 clothianidin’s log Kow value (1.12), 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 clothianidin (a lower bound application rate of 0.066 lb a.i./A with 1 application per year and an upper bound application rate of 0.4 lb a.i./A with 1 application per year) and are provided below in **Table 3-11**. 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 clothianidin and associated application rates are provided in **APPENDIX 1-2**. **Table 3-11** summarizes the mean and upper-bound dietary-based EECs and the associated base model that is used in the MAGtool to predict the EECs. Clothianidin uses also include seed treatment uses; 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 3-11. 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****(0.066 lb a.i./A x 1 application/year)** | **Upper bound application rate****(0.4 lb a.i./A x 1 applications/year)** |
| **Upper Bound** | **Mean** | **Upper Bound** | **Mean** |
| Short Grass | T-REX | 15.8 | 5.6 | 96 | 34 |
| Tall Grass, nectar and pollen | T-REX | 7.3 | 2.4 | 44 | 14.4 |
| Broadleaf plants | T-REX | 8.9 | 3.0 | 54 | 18 |
| Seeds, fruit and pods | T-REX | 1 | 0.5 | 6.0 | 2.8 |
| Arthropods (above ground) | T-REX | 6.2 | 4.3 | 37.6 | 26 |
| Soil-dwelling invertebrates (earthworms) | Earthworm fugacity | 0.009 | NA | 0.056 | NA |
| Small mammals (15 g, short grass diet) | T-HERPS | 15.58 | 5.52 | 94.4 | 33.4 |
| Large mammals (1000 g, short grass diet)2 | T-HERPS | 2.5 | 0.88 | 15.1 | 5.36 |
| Small birds (20 g, insect diet) | T-HERPS | 16.96 | 11.73 | 102.8 | 71.1 |
| Small terrestrial phase amphibians or reptiles (2 g; insect diet) | T-HERPS | 0.83 | 0.57 | 5.01 | 3.47 |
| 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 clothianidin in aquatic dietary items

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2. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling> (accessed January 2020) [↑](#footnote-ref-3)
3. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values> (accessed January 2020) [↑](#footnote-ref-4)
4. The draft guidance is available at www.regulations.gov docket number: EPA-HQ-OPP-2013-0676 [↑](#footnote-ref-5)
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