Chapter 3 – Draft Propazine Exposure Characterization

Contents

[1 Environmental Transport and Fate Characterization 3](#_Toc53015576)

[2 Identification of Transformation Products of Concern 5](#_Toc53015577)

[3 Measures of Aquatic Exposure 6](#_Toc53015578)

[3.1 Aquatic Exposure Models 6](#_Toc53015579)

[3.2 HUC and Use Site Crosswalk 10](#_Toc53015580)

[3.3 Scenario Selection 10](#_Toc53015581)

[3.4 Application Practices 10](#_Toc53015582)

[3.4.1 Application Method 10](#_Toc53015583)

[3.4.2 Spray Drift 10](#_Toc53015584)

[3.4.3 Application Timing 11](#_Toc53015585)

[3.5 Plant Assessment Tool (PAT) 11](#_Toc53015586)

[3.6 Aquatic Modeling Input Parameters 12](#_Toc53015587)

[3.7 Aquatic Modeling Results 13](#_Toc53015588)

[3.8 Available Monitoring Data 14](#_Toc53015589)

[3.8.1 General Monitoring Data 14](#_Toc53015590)

[3.8.1.1 Water Quality Data Portal 15](#_Toc53015591)

[3.9 Aquatic Exposure Summary 15](#_Toc53015592)

[3.10 Uncertainties in Aquatic Modeling and Monitoring Estimates 16](#_Toc53015593)

[3.11 Uncertainties with the Plant Assessment Tool (PAT) 17](#_Toc53015594)

[4 Measures of Terrestrial Exposure 18](#_Toc53015595)

[5 Literature Cited 20](#_Toc53015596)

Tables

[Table 1‑1. General physical-chemical and environmental fate properties of Propazine. 4](#_Toc53021904)

[Table 2‑1. Chemical Names and Structures for Propazine and Transformation Products. 5](#_Toc53021905)

[Table 3‑1. Aquatic Bin, Modeled Waterbody Crosswalk. 9](#_Toc53021906)

[Table 3‑2. Estimated Spray Drift Fractions for Different Aquatic Bins and Application Methods. 11](#_Toc53021907)

[Table 3‑3. Input Values Used for Tier II Surface Water Modeling of Propazine with PWC. 12](#_Toc53021908)

[Table 3‑4. The Range of PWC Daily Average Water Column EECs for Propazine. 13](#_Toc53021909)

[Table 3‑5. The Range of PWC Pore Water EECs for Propazine. 13](#_Toc53021910)

[Table 3‑6. WQP Data Summary by HUC 2 and Number of Samples. 15](#_Toc53021911)

[Table 3‑7. 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. 19](#_Toc53021912)

Figures

[Figure 3‑1. Hydrologic Unit Code (HUC) 2-Digit Regions and Associated Meteorological Data. 8](#_Toc52973126)

[Figure 3‑2. Effect of Pesticide Concentration via Advective Dispersion. 17](#_Toc52973127)

# Environmental Transport and Fate Characterization

Propazine has a high potential to leach into ground water and reach surface waters by runoff and drift during aerial or ground applications. In areas where the soils are highly permeable, the water table is shallow, and sufficient precipitation and/or irrigation occur, the use of propazine may result in ground water contamination. The environmental fate properties of propazine, along with monitoring data identifying its presence in surface waters, indicate that important transport pathways include runoff and spray drift.

Physical-chemical properties and known environmental fate and transport-related properties for propazine are provided in Table 1‑1. Volatility is not expected to be a major route of exposure for propazine due to the low vapor pressure (2.9 x 10-8 torr at 20°C). Based on propazine’s aerobic soil metabolism and aerobic and anaerobic aquatic metabolism data, propazine is considered persistent[[1]](#footnote-2) in the environment, with half-lives on the order of months to years (representative[[2]](#footnote-3) half-life values range from 69 to 1,240 days). Current studies indicate that propazine is stable to hydrolysis and photolysis. However, published literature on propazine and related chloro-s-triazines indicate that the chemical may be susceptible to hydrolysis after adsorption onto the surface of soil colloids -- a surface catalysis effect (Khan, 1980; Wolfe et al, 1991; Russell et al, 1968; Brown and White, 1969; Nearpass, 1972).

Propazine is mobile with Kocs ranging from 65-268 L/kg[[3]](#footnote-4) and Freundlich Kd values from 0.34 to 3.19. The mobility of propazine is also noted in the fields, where supplemental terrestrial field dissipation studies suggest that propazine persists in the upper 6 inches and may leach to ground water. Montgomery (1993) summarized soil adsorption data from four studies (Burkhard and Guth, 1981; Harris, 1966; Talbert and Fletchall, 1965; Walker and Crawford, 1970) involving 38 soils. The reported adsorption Kd values averaged 3.4 mL/g, with a range of 0.1 to 20.5. In 35 of the 38 soils, the Kd values were less than 4.7. The Koc values averaged 155 mL/g (ranging from 29 to 363), which are within the range of the Koc values reported in the above-mentioned environmental fate studies submitted by the registrant. It has also been reported in the literature that if released to soil, propazine will persist longer in dry or cold conditions or other conditions which inhibit biological and chemical activity (Worthing, 1983). Therefore, in areas where soils are highly permeable, the water table is shallow, or where there is irrigation and/or high rainfall, the use of propazine may result in ground water contamination.

Table ‑. General physical-chemical and environmental fate properties of Propazine.

| **Parameter** | **Value** | **Reference / MRID** |
| --- | --- | --- |
| **Selected Physical/Chemical Parameters** | | |
| IUPAC Name | 6-chloro-N2,N4-diisopropyl-1,3,5-triazine-2,4-diamine | USEPA, 2015 |
| Chemical Abstracts Service (CAS) Registry Number | 139-40-2 | USEPA, 2015 |
| Chemical Structure | C:\Users\KCrews\AppData\Local\Microsoft\Windows\INetCache\Content.MSO\9336B37A.tmp | -- |
| Molecular Formula | C9H16ClN5 | USEPA, 2015 |
| Molecular Weight | 229.71 g/mol | USEPA, 2015 |
| Water Solubility | 8.6 mg∙L-1 at 20° C | USEPA, 2015 |
| Vapor Pressure | 2.9 x 10-8 torr at 20° C | USEPA, 2015 |
| Density | 1.16 kg∙L-1 @ 20° C | USEPA, 2015 |
| Henry's Law Constant | 1.02 x 10-9 atm∙m3∙mol-1 (calculated) | USEPA, 2015 |
| Log Octanol/Water Partition Coefficient (Log Kow) | 2.91 | USEPA, 2015 |
| Persistence | | |
| Hydrolysis (t½ @ 25°C) | stable, pH 5  stable, pH 7  stable, pH 9 | 43689802 (acceptable) |
| Aqueous Photolysis (t½ @ 25°C) | stable | 44184805 (acceptable) |
| Soil Photolysis (t½ @ 25°C) | stable | 44184806 (acceptable) |
| Aerobic Soil metabolism (t½) | 423 days (sandy loam, pH 6.8, 25°C) | 44184807 (acceptable) |
| Anaerobic Aquatic metabolism (t½ @ 20°C) | 122 days (Golden Lake Sandy Loam)  69.3 days (Goose River Clay Loam) | 48125803 (supplemental) |
| Aerobic Aquatic metabolism (t½ @ 20°C) | 1240 days (Golden Lake Sandy Loam)  146 days (Goose River Clay Loam) | 48125802 (supplemental) |
| Foliar Degradation (t½) | -- | No data |
| Mobility/Adsorption-Desorption | | |
| Soil –water distribution coefficients (L·kg-1) | Kd =0.34; KOC =83 (loamy sand)  Kd =1.14; KOC =123 (sandy loam)  Kd =2.69; KOC =158 (loam)  Kd =3.19; KOC =65 (clay loam)  Kd =0.67; KOC =268 (sand)  Kd =1.28; KOC =128 (sandy loam)  Kd =1.30; KOC =96 (silty clay)  Kd =1.35; KOC =79 (loam) | 00152997 (acceptable)  43689804 (acceptable) |
| Bioconcentration | Whole fish (bluegill) tissue BCF = 20 | 44184812 (acceptable) |
| Field Dissipation | | |
| Terrestrial Field Dissipation (DT50) | 51.7 days, total propazine residue  (TX, loamy sand) | 44184809 (supplemental) |

# Identification of Transformation Products of Concern

Table 2‑1 gives the chemical names and structures for all atrazine transformation products. No studies were submitted on the persistence of degradates (both major and minor) in the environment. The major residue of concern is parent propazine, only. Under laboratory aerobic soil conditions, the major soil metabolite was 2-hydroxy propazine (2-hydroxy-4,6-bis(isopropy1amino)-s-triazine) and comprised a maximum of 31% of the total applied radioactivity after one year. Minor degradates consist of desethylatrazine (2-amino-4-chloro-6-isopropylamino-s-triazine or DEA) (<2% of the total applied radioactivity) and 2-hydroxy desethylatrazine (<5% of the total applied radioactivity).

Five transformation products were identified in aerobic and anaerobic aquatic studies: 2-chloro-4,6-diamino-1,3,5-triazine (DACT); atrazin-desethyldeisopropyl-2-hydroxy (ammeline); propazine-2-hydroxy (2-hydroxy propazine); atrazin-desethyl-2-hydroxy (deisopropyl hydroxy propazine); and atrazin-desethyl (DEA). In the aerobic aquatic study, DACT was a major transformation product (maximum of 11.7% at Day 46) in the Golden Lake water-loamy sand sediment system and a minor transformation product in the Goose River water-clay loam sediment system. All other compounds were minor transformation products in both systems. In the anaerobic aquatic study, no major transformation products were identified. All identified transformation products were minor. No major transformation products were isolated and identified from either system; however, major products may have been isolated but not identified.

The major laboratory soil degradate, 2-hydroxy propazine was seen in the 0-3” and 3-6” soil layers of the terrestrial field studies at approximately 15% of parent at day 1 and decreased to less than 5% of parent by day 93. The other two minor degradates desethylatrazine (DEA) and 2,4-diamino-6 chloro-s-triazine (DACT), which are common to atrazine and simazine, were detected only in the 0-3" soil layer, each at less than 5% of parent at day 1, however decreasing to less than 1% by day 28.

An analysis of the residues of concern is provided in **APPENDIX 1-8**. Based on the analysis of formation and toxicity data on the known transformation products, aquatic modeling of the parent compound alone for each of the triazines is considered adequate for determining potential exposure concentrations to aquatic organisms. The risks associated with the degradates are not expected to materially influence (i.e., change) any species-specific effects determination that is based on propazine parent.

Table ‑. Chemical Names and Structures for Propazine and Transformation Products.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Name** | **Chemical Name** | **Chemical Formula** | **CAS Reg No.** | **Structure** |
| Propazine (parent) | 6-chloro-N2,N4-diisopropyl-1,3,5-triazine-2,4-diamine | C9H16ClN5 | 139-40-2 |  |
| 2-hydroxy propazine | 2-hydroxy-4,6-bis(isopropy1amino)-s-triazine | C9H17ON5 | 7374-53-0 |  |
| Desethylatrazine (DEA) | 2-amino-4-chloro-6-isopropylamino-s-triazine | C6H10ClN5 | 6190-65-4 |  |
| 2-hydroxy desethylatrazine | Deisopropyl hydroxy propazine | C6H10N5O | 19988-24-0 |  |
| Diadealkylatrazine (DACT) | 2-chloro-4,6-diamino-1,3,5-triazine | C3H4ClN5 | 3397-62-4 |  |
| Ammeline | atrazin-desethyldeisopropyl-2-hydroxy | C3H5N5O | 645-92-1 |  |

# Measures of Aquatic Exposure

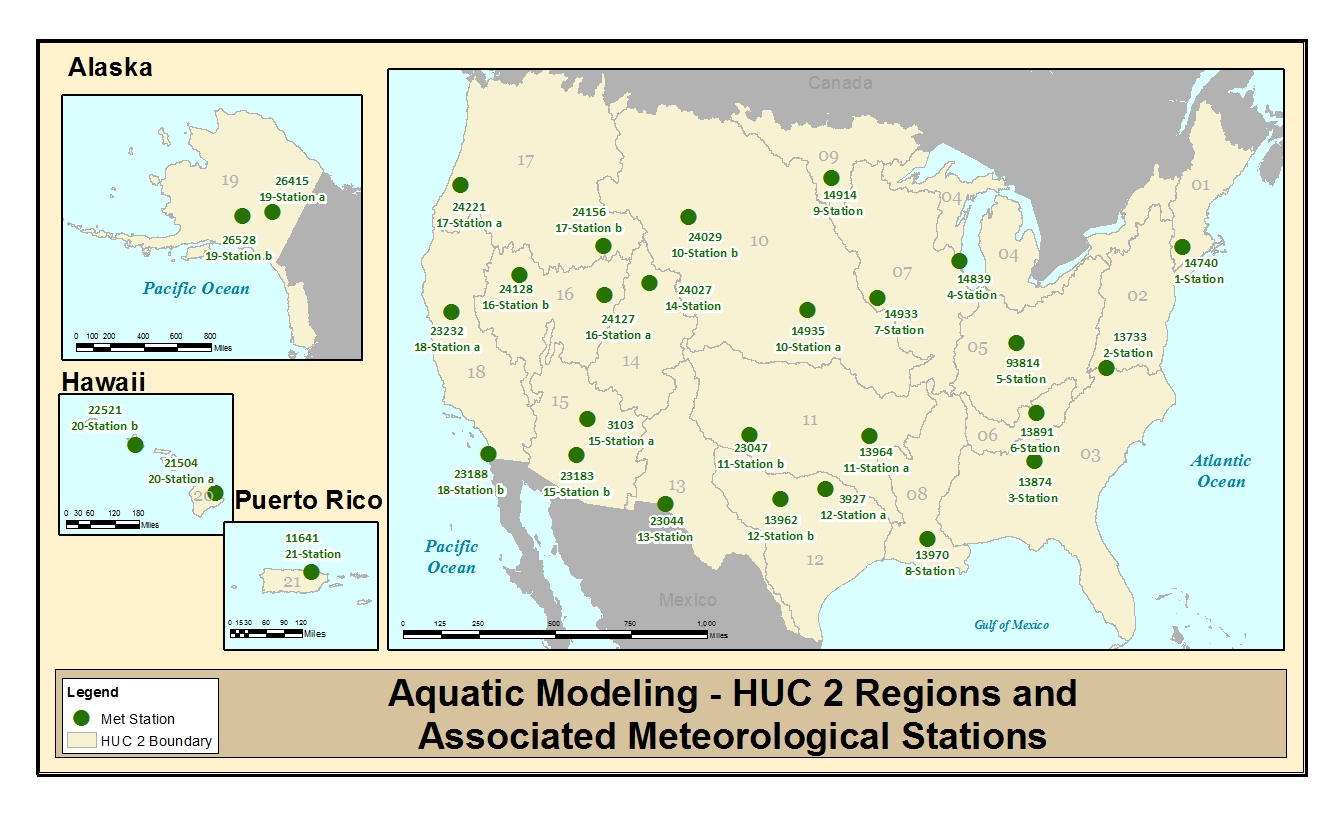
Maximum application rates and minimum application retreatment intervals are modeled to estimate the exposure to propazine based on the scenario development document (**APPENDIX 1-3**) developed for propazine, unless otherwise noted.

Propazine-specific modeling scenarios are used for modeling each use. This includes the selection of scenarios and agronomic practices (*e.g.*, applications methods, dates). Table 3‑3 includes model input parameters as well as the justification for selecting these parameters; however, the general approaches used are described below.

Aquatic Exposure Models

Aquatic exposures (surface water and benthic sediment pore water) are quantitatively estimated for representative propazine uses included on the scenario development document (**APPENDIX 1-3**) by HUC 2 Regions (Figure 3‑1) and by aquatic bin (Table 3‑1) using the Pesticide Root Zone Model (PRZM5) and the Variable Volume Water Model (VVWM)[[4]](#footnote-5) 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 the 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 was not expected that this assumption was 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.

Figure ‑. Hydrologic Unit Code (HUC) 2-Digit Regions and Associated Meteorological Data.

For propazine, 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 3‑1 provides a crosswalk of the bins and how they were modeled. While the standard farm pond is bigger than bin 6, the EECs estimated for bin 6 in previous BEs were similar 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.

For bin 1, a wetland the size of the standard farm pond (1 ha) is modeled receiving runoff from a 10-ha field. The depth of the wetland varies from 0.5 to 15 cm, simulating the potential for the wetland to fill up and dry down, and is simulated in PRZM/VVWM using the varying volume and flow through model. The sediment layer is increased from the standard 5 cm to 15 cm to represent the typical active root zone for wetland species of forbs and woody plants. The wetland is simulated in PRZM/VVWM using the variable volume. The results of the wetland modeling are then processed in the Plant Assessment Tool (**Section 3.5**).

Table ‑. 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.

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

HUC and Use Site Crosswalk

The National Agricultural Statistics Census of Agriculture 2012 (NASS) data along with the Cropland Data Layer (CDL) were used to determine which crops would be modeled within each represented HUC 2. Additionally, specific geographic limitations on how a product may be applied to particular crops were considered when determining what rates would be simulated for different HUC 2 regions. For propazine, the technical registrant has submitted an Endangered Species Act commitment letter to limit the application of propazine on sorghum to the states of Texas, Oklahoma, and Kansas. As such, only HUC 2 regions 10, 11, 12, and 13 are being modeled. A crop use layer-HUC 2 Region matrix for propazine is provided in **APPENDIX 3-1**.

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 **APPENDIX 1-3**.

Application Practices

Application Method

During application of pesticides, methods of application as well as product formulation (e.g., liquid) used by an applicator can impact the off-site transport of the active ingredient. Label directions (such as spray drift buffers, 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 two types of propazine applications included in the scenario development document (**APPENDIX 1-3**). Applications can be made via ground sprayer or aerial broadcast.

Spray Drift

Propazine has one flowable formulation and the spray drift fractions used for all applications in PWC modeling are shown in Table 3‑2. Drift fractions were derived using AgDRIFT Tier 1 models and default spray droplet spectra (Fine to Medium for aerial applications and Very Fine to Fine, high boom for ground applications.

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

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Aquatic Bin** | **Buffer (m)** | **Waterbody** | | **Spray Drift Fraction**  **(unitless)** | |
| **Depth (m)** | **Width (m)** | **Aerial** | **Ground** |
| 1 | 0 | 0.15 | 64 | 0.1254 | 0.0616 |
| 41 | 20 | 2.74 | 82 | 0.071 | 0.0185 |
| 7 | 61 | 2 | 64 | 0.0314 | 0.0078 |

1 Drift fraction for this waterbody also takes into account spray drift entering a 4 m wide stream which enters the waterbody.

Application Timing

In selecting application dates for aquatic modeling, EPA considers a number of 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.

Propazine applications may occur before planting or after planting but before sorghum or weeds emerge for control of annual broadleaf weeds. PWC model inputs for the application dates were chosen based on these timings and were selected to represent conservative and reasonable estimates. For details on application date selection for use of propazine, see **APPENDIX 1-3** and **APPENDIX 3-1**.

Plant Assessment Tool (PAT)

The Plant Assessment Tool (PAT) employs mechanistic representations of fate (e.g., degradation) and transport (e.g., runoff), using data that are typically available for pesticides, to model runoff and spray drift exposure to terrestrial and wetland environments. 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.

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 propazine is presented in Table 3‑3. 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[[5]](#footnote-6) (USEPA, 2009),
* *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[6]](#footnote-7)* (NAFTA, 2012; USEPA, 2012b), and
* *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[7]](#footnote-8) (USEPA, 2013)

Table ‑. Input Values Used for Tier II Surface Water Modeling of Propazine with PWC.

| **Parameter (units)** | **Value (s)** | **Source** | **Comments** |
| --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (KOC (L/kg-OC)) | 125 | MRIDs 00152997  43689804 | Average of eight KOC estimates from eight soils ranging from 65 to 268 L/kg-OC. Koc better predictor of sorption based on lower CV. |
| Water Column Metabolism Half-life or Aerobic Aquatic Metabolism Half-life (days) and Reference Temperature | 2380 at 20°C | MRID 48125802 | Represents the 90th percent upper confidence bound on the mean of two representative half-life values. Half-life values are representative of aerobic aquatic metabolism ranging from 146 to 1240 days. |
| Benthic Metabolism Half-life or Anaerobic Aquatic Metabolism Half-life (days) and Reference Temperature | 177 at 20°C | MRID 48125803 | Represents the 90th percent upper confidence bound on the mean of two representative half-life values. Half-life values are representative of anaerobic aquatic metabolism ranging from 69.3 to 122 days. |
| Aqueous Photolysis Half-life @ pH 7 (days) and Reference Latitude 40° N latitude, 25°C | 0 | MRID 44184805 | Degradation occurring in photolysis study under 24-hour irradiation xenon arc lamp; 211 days dark control. |
| Hydrolysis Half-life (days) | 0 | MRID 43689802 | stable |
| Soil Half-life or Aerobic Soil Metabolism Half-life (days) and Reference Temperature | 1269 at 25oC | MRID  44184807 | Representative half-life value (423 days) from one study times three. |
| MWT or Molecular Weight (g/mol) | 229.71 g/mol | USEPA, 2015 | -- |
| Vapor Pressure (Torr) at 30°C | 2.90 × 10-8 | USEPA, 2015 | -- |
| Solubility in Water @ 20°C, pH not reported (mg/L) | 8.6 mg/L | USEPA, 2015 | -- |
| Henry's Law Constant (atm∙m3∙mol-1) | 1.02 x 10-9 | USEPA, 2015 | calculated |
| Foliar Half-life (days) | 0 | -- | No data, assume no degradation on foliage |
| Application Efficiency | Aerial: 0.95  Ground: 0.99 | Input parameter guidance (USEPA, 2009) | -- |
| Drift | Estimated | AgDRIFT | See **Section 3.4.2** |

The coefficient of variation for propazine was 52% for KOC and 65% for Kd based on the values presented Table 3‑3. The KOC was chosen as a better predictor of sorption based on a lower coefficient of variation (CV). The CV expresses (as a percent) the level of dispersion of the given values around the mean for each parameter. Hence, the KOC has a lower amount of variability amongst the given values.

Aquatic Modeling Results

The estimated environmental concentrations (EECs) derived from the PWC modeling based on maximum labeled rates included in the master use summary document, by HUC 2, are summarized for the various aquatic bins in Table 3‑4 and Table 3‑5, for water column and pore water, respectively. 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 ‑. The Range of PWC Daily Average Water Column EECs for Propazine.

| **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 10a | 1296 – 2631 | 265 – 277 | 43.6 – 44.8 | 43.6 – 44.8 | 265 – 277 | 63.4 – 68.7 | 63.4 – 68.7 |
| HUC 10b | 1302 – 2641 | 280 – 292 | 39.6 – 45.2 | 39.6 – 45.2 | 280 – 292 | 36.4 – 43.9 | 36.4 – 43.9 |
| HUC 11a | 1220 – 2465 | 607 – 633 | 90 – 88.6 | 90 – 88.6 | 607 – 633 | 119 – 119.8 | 119 – 119.8 |
| HUC 11b | 1297 – 2632 | 559 – 582 | 67 – 68.7 | 67 – 68.7 | 559 – 582 | 64.5 – 68.9 | 64.5 – 68.9 |
| HUC 12a | 1291 – 2624 | 540 – 563 | 89.1 – 90.5 | 89.1 – 90.5 | 540 – 563 | 135.7 – 135.8 | 135.7 – 135.8 |
| HUC 12b | 1293 – 2626 | 540 – 563 | 83.3 – 84.3 | 83.3 – 84.3 | 540 – 563 | 98 – 99.3 | 98 – 99.3 |
| HUC 13 | 1293 - 2627 | 265 – 276 | 31.4 – 37.4 | 31.4 – 37.4 | 265 – 276 | 17.4 – 22.1 | 17.4 – 22.1 |

Table ‑. The Range of PWC Pore Water EECs for Propazine.

| **HUC 2** | **Range of 1-in-15 year Pore Water EECs (µg/L)** | | | | | | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Bin 1** | **Bin 2\*** | **Bin 3** | **Bin 4** | **Bin 5\*** | **Bin 6** | **Bin 7** |
| HUC 10a | 26.8 – 36.6 | 279 – 291 | 37.9 – 39.8 | 37.9 – 39.8 | 279 – 291 | 58.6 – 62.5 | 58.6 – 62.5 |
| HUC 10b | 29.2 – 41.9 | 298 – 311 | 39.5 – 44.8 | 39.5 – 44.8 | 298 – 311 | 31.3 – 38 | 31.3 – 38 |
| HUC 11a | 26.6 – 29.5 | 609 – 634 | 52.5 – 53.3 | 52.5 – 53.3 | 609 – 634 | 104.2 – 105 | 104.2 – 105 |
| HUC 11b | 38.5 – 45.6 | 561 – 585 | 47.4 – 47.7 | 47.4 – 47.7 | 561 – 585 | 53.9 – 57.7 | 53.9 – 57.7 |
| HUC 12a | 32.1 – 37.6 | 543 – 565 | 64.5 – 66.1 | 64.5 – 66.1 | 543 – 565 | 111.3 | 111.3 |
| HUC 12b | 31.4 – 35.9 | 543 – 566 | 67 – 67.04 | 67 – 67.04 | 543 – 566 | 83 – 84.3 | 83 – 84.3 |
| HUC 13 | 17.8 – 24.5 | 266 – 277 | 22.3 – 26.6 | 22.3 – 26.6 | 266 – 277 | 14.9 - 18 | 14.9 - 18 |

\*Pore water concentrations for bins 2 and 5 have been estimated using edge-of-field runoff from the dissolved and eroded soil amounts from the ZTS file.

Currently EFED models flowing waterbodies in its FIFRA assessments using the index reservoir, a waterbody with a surface area of 5.26x105 m2 and a depth of 2.74 m, with a watershed area of 1.728x106 m2. This conceptual model is based upon the Shipman Reservoir in Illinois and has been reviewed by EPA’s FIFRA SAP. Although the dimensions are smaller than those of a bin 3 or 4 and the flowrate is much lower, EFED has confidence in the estimated concentrations developed using the index reservoir and believes these EECs can be used as a conservative surrogate for bins 3 and 4 until EFED can validate an alternative conceptual model for bins 3 and 4.

Available Monitoring Data

General Monitoring Data

Examination of the EPA 303(d) list of impaired waters[[8]](#footnote-9) for pesticides indicates no impairments caused by parent propazine, as of June 25, 2020.

Water monitoring data were obtained from the National Water Quality Monitoring Council’s Water Quality Data Portal ([USEPA and USGS, 2013](#_ENREF_36)), 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 propazine use occurs. Also, the sampling sites, as well as the number of samples, vary by year. The vulnerability of the sampling site to propazine contamination varies substantially due to use, soil characteristics, weather and agronomic practices. None of the monitoring programs examined to date were specifically designed to target propazine use. Therefore, peak concentrations of propazine likely went undetected in these programs. The various monitoring programs did not detect propazine with high frequency, but propazine detections ranged from 0.0005 µg/L up to 13 µg/L. Many of the high detections were historical, but the extent to which historical values represent current agronomic or labeled use instructions is uncertain.

Therefore, while there are many individual samples collected and analyzed for propazine 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 propazine.

Water Quality Data Portal

Comprehensive surface and ground water propazine data were obtained on June 3, 2020 in a download of data from the Water Quality Data Portal (<http://www.waterqualitydata.us/>). Table 3‑6 summarizes the WQP data and number of samples for propazine in the various HUC 2 regions (10, 11, 12, and 13) corresponding to the states where propazine is applied. The years with reported data range from 1979 to 2020 with a total of 2,293 number of sites for all four areas of interest, combined. Of the combined 31,613 samples, 93% were reported as “no detections.” The reported detections ranged from 0 μg/L to 13 μg/L, with HUC 2 region 11 having the highest reported detection.

Table ‑. WQP Data Summary by HUC 2 and Number of Samples.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **HUC2** | **Years** | **Number of Sites** | **Number of Samples** | **Number of No Detections** | **Range of Detections (μg/L)** |
| 10 | 1981 - 2020 | 1394 | 20080 | 18576 | 0 - 4.9 |
| 11 | 1979 - 2020 | 752 | 10154 | 9798 | 0 - 13 |
| 12 | 1981 - 2020 | 97 | 1041 | 841 | 0 - 2.1 |
| 13 | 1986 - 2019 | 51 | 338 | 334 | 0 – 0.01 |

Aquatic Exposure Summary

Model-derived EECs represent an upper bound on potential exposure as a result of the use of propazine. 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 roughly the same order of magnitude or an order of magnitude greater than the measured concentrations. The medium and high flowing and static bins were used for comparison purposes as these would seem to represent typical waterbodies considered for ambient water monitoring, as they typically have flow and water present all year. As recommended by the NRC in the 2013 NAS report, general monitoring data are not recommended to be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of EPA’s fate and transport models. However, EPA believes monitoring data can be used as part of the weight-of-evidence evaluation to evaluate potential exposure.

Uncertainties in Aquatic Modeling and Monitoring Estimates

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

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

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

The assessment relies on maximum use patterns (**APPENDIX 1-3**). 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 taken into account.

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 taken into account 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.

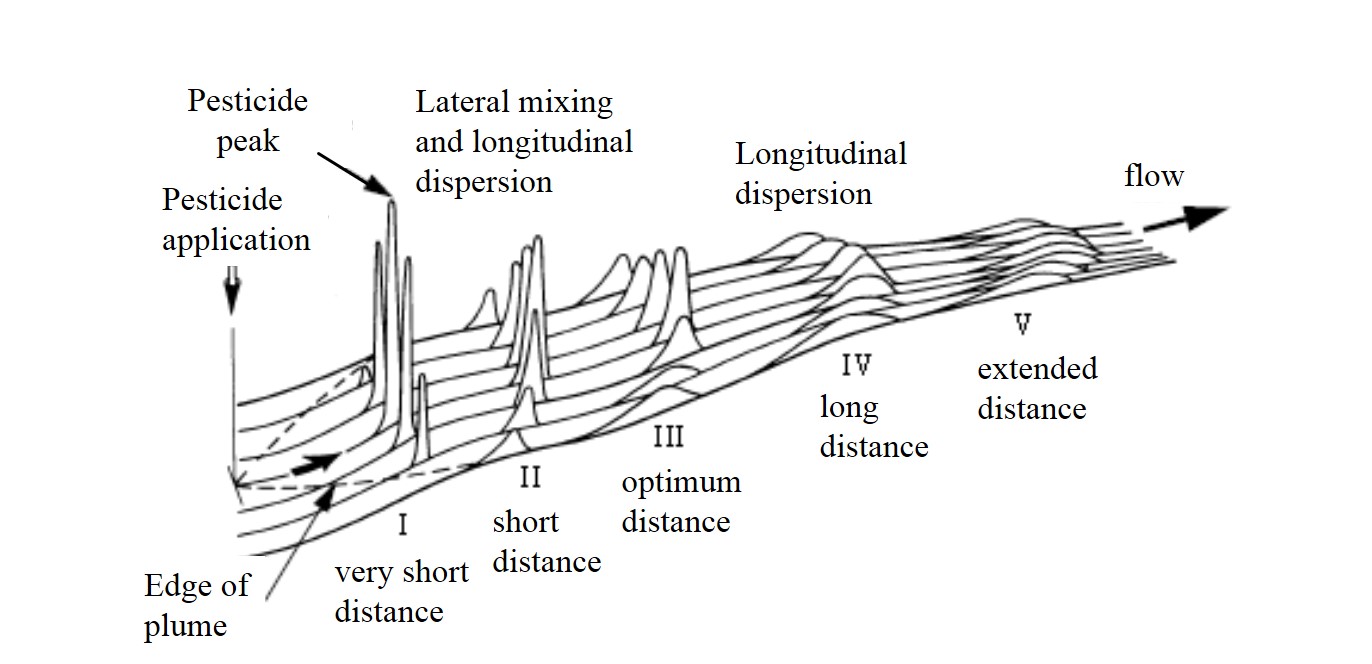


Figure ‑. Effect of Pesticide Concentration via Advective Dispersion.

Uncertainties with 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; etc.) 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][[9]](#footnote-10)). 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 propazine 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). 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. A default foliar dissipation half-life of 35 days is used for propazine, based on data reported by Willis and McDowell (1987).

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, 2016d), 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**).

Potential terrestrial EECs used in the effects determinations were calculated for the 1.2 lbs a.i./A application rate for propazine and are provided below in Table 3‑7. 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). Additionally, other information considered in Step 2 (*e.g.,* typical use rates, use rates based on maximum usage in a species range, distribution of EECs, etc.), could alter the EECs used to assess a species exposure.Table 3‑7summarizes the mean and upper bound dietary-based EECs and the associated base model that is used in the MAGtool to predict the EECs.

Table ‑. 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.

|  |  |  |  |
| --- | --- | --- | --- |
| **Food Item** | **Model** | **Maximum application rate**  **1.2 lbs a.i./A x 1 application/year)** | |
| **Upper Bound** | **Mean** |
| Short Grass | T-REX | 288 | 102 |
| Tall Grass, nectar and pollen | T-REX | 132 | 43 |
| Broadleaf plants | T-REX | 162 | 54 |
| Seeds, fruit and pods | T-REX | 18 | 8.4 |
| Arthropods (above ground) | T-REX | 113 | 78 |
| Soil-dwelling invertebrates (earthworms) | Earthworm fugacity | 19 | NA1 |
| Small mammals (15 g, short grass diet) | T-HERPS | 275 | 97 |
| Large mammals (1000 g, short grass diet)2 | T-HERPS | 44 | 16 |
| Small birds (20 g, insect diet) | T-HERPS | 128 | 89 |
| Small terrestrial phase amphibians or reptiles (2 g; insect diet) | T-HERPS | 6.3 | 4 |
| 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 BCF=20

Literature Cited

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

Brown, C.B., and J.L. White. 1969. Reactions of 1,2-s-triazines with soil clays. Soil Sci. Soc.

Am. Proc. 33:863-867.

Burkhard, N., and J.A. Guth. 1981. Chemical hydrolysis of 2-chloro-4,6-bis(alky1-amino)-1,3,5-triazine

herbicides and their breakdown under the influence of adsorption. Pestic. Sci. 12:45-52.

FAO. 2000. Appendix 2. Parameters of pesticides that influence processes in the soil. In FAO Information Division Editorial Group (Ed.), *Pesticide Disposal Series 8. Assessing Soil Contamination. A Reference Manual*. Rome: Food & Agriculture Organization of the United Nations (FAO). Available at <http://www.fao.org/DOCREP/003/X2570E/X2570E06.htm>.

Harris, C.I. 1966. Adsorption, movement, and phytotoxicity of monuron and s-triazine herbicides in soils. Weeds 14:6-10.

Jones, R.D., Abel, S., Effland, W., Matzner, R., Parker, R. 1998. An Index Reservoir for Use in Assessing Drinking Water Exposure. [https://archive.epa.gov/scipoly/sap/meetings/web/html/](https://archive.epa.gov/scipoly/sap/meetings/web/html/072998_mtg.html)

[072998\_mtg.html](https://archive.epa.gov/scipoly/sap/meetings/web/html/072998_mtg.html).

Khan, Shahamat U. 1980. Pesticides in the soil environment. Elsevier Scientific Publishing

Company, Amsterdam. p. 90-97.

Montgomery, J.H. 1993. Agrochemicals desk reference: environmental data. Lewis Publishers, Chelsea, MI. p. 356-358.

NAFTA. 2012. *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media*. December 2012. NAFTA Technical Working Group on Pesticides. Available at <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values>.

Nearpass, D.C. 1972. Hydrolysis of propazine by the surface acidity of organic matter. Soil Sci. Soc. Am. Proc. 36:606-610.

Russell, J.D., M. Curz, and J.L. White. 1968. Mode of chemical degradation of s-triazines by

montmorillonite. Science 160: 1340- 1342.

64 FR 60194. Federal Register. 1999. Category for Persistent, Bioaccumulative, and Toxic New Chemical Substances. 64:213. Available at <https://www.govinfo.gov/content/pkg/FR-1999-11-04/pdf/99-28888.pdf>.

Talbert, R.E., and O.H. Fletchall. 1965. The adsorption of some s-triazines in soils. Weeds 13:46-52.

USEPA. 2009 *Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides, Version 2.1*. Environmental Fate and Effects Division. Office of Pesticide Programs. U.S. Environmental Protection Agency. Available at https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling.

USEPA. 2015. *Preliminary Risk Assessment for Registration Review of Propazine*. October 28, 2015. Environmental Fate and Effects Division. Office of Pesticide Programs. U. S. Environmental Protection Agency.

USEPA. 2012b. *Standard Operating Procedure for Using the NAFTA Guidance to Calculate Representative Half-life Values and Characterizing Pesticide Degradation*. November 30, 2012. Environmental Fate and Effects Division. Office of Pesticide Programs. U.S. Environmental Protection Agency. Available at <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values>.

USEPA. 2013. *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessment*. Environmental Fate and Effects Division. Office of Pesticide Programs. Office of Chemical Safety and Pollution Prevention. U.S. Environmental Protection Agency. Available at <http://www.regulations.gov/#!docketDetail;D=EPA-HQ-OPP-2013-0676>.

USEPA. 2016a. *Biological Evaluation for Chlorpyrifos Endangered Species Assessment*. March 31, 2016. Environmental Fate and Effects Division. Office of Pesticide Programs. U. S. Environmental Protection Agency. Available at <https://www.epa.gov/endangered-species/biological-evaluation-chapters-chlorpyrifos-esa-assessment>.

USEPA. 2016b. *Biological Evaluation for Diazinon Endangered Specis Assessment*. March 31, 2016. Environmental Fate and Effects Division. Office of Pesticide Programs. U.S. Environmental Protection Agency. Available at <https://www.epa.gov/endangered-species/biological-evaluation-chapters-diazinon-esa-assessment>.

USEPA. 2016c. *Biological Evaluation for Malathion Endandered Species Assessment*. March 31, 2016. Environmental Fate and Effects Division. Office of Pesticide Programs. U. S. Environmental Protection Agency. Available at <https://www.epa.gov/endangered-species/biological-evaluation-chapters-malathion-esa-assessment>.

USEPA. 2016d. Provisional Models for Endangered Species Pesticide Assessments. Environmental Fate and Effects Division. Office of Pesticide Programs. U.S. Environmental Protection Agency. Available at <https://www.epa.gov/endangered-species/provisional-models-endangered-species-pesticide-assessments#Terrestrial>.

USEPA, & USGS. 2013. *Water Quality Portal*. United States Environmental Protection Agency. United States Geological Survey. Available at [http://www.waterqualitydata.us/portal.jsp#](http://www.waterqualitydata.us/portal.jsp).

Walker, A., and D.V. Crawford. 1970. Diffusion coefficients for two triazine herbicides in six soils. Weed Res. 10: 126-132.

Wolfe, N.L., U. Mingelgrin, and G.C. Miller. 1990. Abiotic transformations in water, sediments, and soil. p. 103-1 68 In H.H. Cheng (ed.) Pesticides in the soil environment: processes, impacts, and modeling. SSSA Book Series No. 2. Soil Science Society of America, Madison, WI.

Worthing, C.R., ed. 1983. The pesticide manual: A world compendium. Croydon, England: The British Crop Protection Council.

Young, D., Fry, M. 2016. PRZM5 A Model for Predicting Pesticides in Runoff, Erosion, and Leachate, Revision A. USEPA/OPP734S16001. May 12, 2016.

1. Based on the Persistent, Bioaccumulative, and Toxic New Chemical Substances classification where half-lives greater than 30 days in water, soil, and sediment are considered persistent (64 FR 60194). [↑](#footnote-ref-2)
2. Half-live values were recalculated using the North American Free Trade Agreement (NAFTA) guidance in estimating degradation kinetics (NAFTA, 2012; USEPA, 2012b). [↑](#footnote-ref-3)
3. Mobility was classified using the Food and Agriculture Organization (FAO) classification system (FAO, 2000) and supplemental sorption coefficients. [↑](#footnote-ref-4)
4. The exposure models can be found at: <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-5)
5. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling> (accessed October 2020) [↑](#footnote-ref-6)
6. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values> (accessed October 2020) [↑](#footnote-ref-7)
7. The draft guidance is available at www.regulations.gov docket number: EPA-HQ-OPP-2013-0676 [↑](#footnote-ref-8)
8. <https://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T> [↑](#footnote-ref-9)
9. Young, D., Fry, M. 2016. PRZM5 A Model for Predicting Pesticides in Runoff, Erosion, and Leachate, Revision A. USEPA/OPP734S16001. May 12, 2016 [↑](#footnote-ref-10)