**ATTACHMENT 3-1. Background Document: Aquatic Exposure Estimation for Endangered Species**

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# Executive Summary

In 2017, the U.S. Environmental Protection Agency (EPA), Office of Pesticide Programs, in partnership with the Fish and Wildlife Service, the National Marine Fisheries Service, and the U.S. Department of Agriculture, developed methods for estimating pesticide exposure concentrations in potentially vulnerable surface water bodies to be used in the assessment of adverse effects to Federally endangered and threatened species and designated critical habitat. Aquatic exposure estimates were generated based on key fate and transport processes, using chemical and application information, soil parameters, and watershed and water body characteristics. This method was used to develop Biological Evaluations (BEs) for chlorpyrifos, diazinon, and malathion. Since then, EPA has revised the methods to incorporate refined use information, streamlined modeling, analysis to support probabilistic modeling, and the use of monitoring data to evaluate downstream transport of pesticides. Recommendations for improving upon these revised methods from stakeholders, the scientific community, and the public are welcome and encouraged.

# Introduction

Methods and modeling techniques have been developed to estimate pesticide aquatic exposure concentrations for endangered species for use in the biological evaluation (BEs). The resources and approaches presented are based on current, well-established surface water modeling tools and provide a foundation for current and future endangered species BEs. As new information becomes available, these tools will continue to be updated and developed. This supporting information is being made available to the public to improve transparency and understanding.

Aquatic exposure assessments are conducted for pesticide registration and registration review under the Federal Insecticide, Fungicide, and Rodenticide (FIFRA) and the Federal Food, Drug, and Cosmetic Act (FFDCA), to determine whether pesticides that are applied to land according to their label can result in water concentrations that may adversely impact human health or aquatic organisms. Aquatic modeling is used to estimate pesticide concentrations in water based on a combination of soil, weather, hydrology, and management/crop use conditions that are expected to maximize the potential for pesticide movement into water. If aquatic exposures are less than the various toxicity endpoints of concern, it may be concluded that the pesticide is unlikely to pose adverse effects to the exposed species (*e.g*., amphibians, fish, invertebrates) based on its labeled uses. In situations where estimated exposures exceed toxicity endpoints, further characterization of the potential exposure and effects is needed. For endangered species, similar methods are used to estimate aquatic exposure; however, certain refinements to the assumed conditions and aquatic exposure pathways (water bodies) are incorporated into the analysis. The following sections discuss methods used to model estimated aquatic exposures occurring in different types of watersheds where endangered species occur and to characterize modeled exposure values based on available monitoring data.

# Aquatic Modeling

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## Surface Water Modeling

Currently the Pesticide Root Zone Model (PRZM5) (Young and Fry, 2014)[[1]](#footnote-1) and the Variable Volume Water Model (VVWM) (Young, 2014)[[2]](#footnote-2) are used to estimate pesticide movement and transformation on an agricultural field and in receiving water bodies, respectively. These models are linked with a user interface, the Pesticide in Water Calculator (PWC). Standard crop-specific scenarios are used to represent combinations of soil, crop, weather, and hydrological factors that are expected to contribute to high-end pesticide concentrations in water.

### Traditional Approach

PRZM5 simulates pesticide sorption to soil, in-field decay, erosion, and runoff from an agricultural field or drainage area following pesticide application(s). The VVWM estimates water and sediment concentrations in an adjacent surface water body receiving the pesticide loading by runoff, erosion, and spray drift from the field. For the endangered species assessments, PRZM5 and VVWM are applied to simulate a range of regionally-specific conditions where endangered species and designated critical habitat may occur. The PRZM5 and VVWM documentation, installation files, and source code are available at the USEPA Water Models website[[3]](#footnote-3). Historically for ecological assessments, the estimated 1-in-10-year return frequency concentrations from the model, for either single-day (peak concentration for estimating acute exposures) or time-averaged periods (for estimating chronic exposures) is compared to relevant toxicity endpoints of concern. This approach is intended to screen out pesticides (and/or specific uses) that are not likely to be of potential concern, and to focus resources on characterizing the exposure to pesticides that exceed the level of concern.

### Revised Conceptual Model and Approach for ESA

Building upon the existing ecological exposure modeling framework (**Section 3.1.1**), the modified approach for ESA delineates additional water body types (or habitats) to characterize a range of potential exposures to endangered species. **Figure 1** (**Table 1**) summarizes the various aquatic habitat bins that are considered in the evaluation of exposure in static and flowing freshwater bodies and estuarine/marine water bodies.

For the ESA Biological Evaluations, 1-in-15-year exposure concentrations are estimated using the daily time series of estimated concentrations from 30-year PRZM5/VVWM simulations, instead of 1-in-10-year concentrations as in traditional ecological exposure assessments. The 1-in-15-year concentrations are used here for consistency with the length of the action (15 years), based on the registration review cycle.



Figure 1. Conceptual model for estimating the aquatic exposure of endangered species to pesticides.

The applied pesticide from edge of the treated field is received by ten potential aquatic habitat bins (static, flowing, estuarine/marine), and estimated exposure concentrations are calculated.

Table 1. Endangered species aquatic habitat bins

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Generic Habitat | Depth (meters) | Width (meters) | Length (meters) | Flow (m3/second) |
| 1 – Aquatic-associated terrestrial habitats1 | 0.005-0.15 | 64 | 156 | 0 |
| 2- Low-flow | 0.1 | 2 | length of field2  | 0.001  |
| 3- Moderate-flow | 1 | 8 | length of field  | 1  |
| 4- High-flow | 2 | 40 | length of field  | 100  |
| 5 – Low-volume | 0.1 | 1 | 1 | 0 |
| 6- Moderate-volume | 1 | 10 | 10 | 0 |
| 7- High-volume | 2 | 100 | 100 | 0 |
| 8- Intertidal near shore | 0.5 | 50 | length of field | NA |
| 9- Subtidal near shore | 5 | 200 | length of field | NA |
| 10- Offshore marine | 200 | 300 | length of field | NA |

1 Dimensions were not defined, as they were for the other 9 bins, for Bin 1. For the purposes of modeling plant exposures in wetlands, dimensions similar to EPA’s standard farm pond were used and reported here.

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

NA – not applicable

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###  Modeling Components

#### Input Scenarios

For aquatic exposure assessments, input “scenarios” are used as a finite set of combinations of soil, weather, hydrology, and management/crop use conditions that are expected to maximize the potential for pesticides to move into surface water. Over the years, EPA has developed a large suite of surface water scenarios (123 total) for use in PRZM5/VVWM simulations, spanning a range of agricultural and non-agricultural pesticide use sites[[4]](#footnote-4). The locations of the existing scenarios are shown in **Figure 2**. However, there are many instances when a scenario does not exist for a particular use (*e.g.,* kiwi fruit), or for the full range of crop use at the national scale.

When a crop use pattern does not have an existing scenario, the use is typically modeled with a surrogate scenario using one of two approaches. In the first approach, the scenario is modeled based on an existing scenario that is representative of that use pattern. This typically entails making the determination that the crop is agronomically similar to the existing scenario (*i.e*., the surrogate crop is grown in a geographic region similar to the crop without a scenario, emerges, matures and is harvested at roughly the same time as the surrogate crop, and has a runoff curve number [an empirical parameter used to predict direct runoff] of similar magnitude). For example, the California almond scenario can be used to model pesticide applications to pistachios.

The second example occurs when a scenario(s) exists, but there are gaps at the national scale relative to the full geographic breadth of the use pattern. In this case, an existing scenario may be modified with a weather station other than that specified in the original scenario file (see **Section 3.1.3.1.4** for more information on weather stations). Because the runoff curve number is fairly generic (USDA, 1986 Tables 2.2a, b and c[[5]](#footnote-5)), holding all chemical inputs the same, a scenario modeled with another weather station can provide a reasonable estimate of exposure relative to the original scenario, by accounting for variations in rainfall and evaporation (*i.e.*, rainfall totals, timing and intensity).

For the evaluation of listed species, a matrix was developed to assign one input scenario per hydrologic unit code 2 (HUC2) region and crop group combination (**Table 2**). The following steps were completed to select the representative scenario (including the weather station) for each HUC2 region-crop group combination.

Figure 2. Location of existing aquatic exposure modeling scenarios

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3. 1. 1.
		2.
		3. 1.

##### Regional Spatial Delineation for Scenarios

HUC2 regions were used as the geospatial reference for scenario selection (**Figure 3**).



Figure 3. Spatial distribution of HUC2 regions and U.S. state boundaries

##### Association to Agricultural and Nonagricultural Data Layers

The crop group for each scenario was based on the USDA National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL)[[6]](#footnote-6), which offers annual, geospatially referenced crop-specific land cover information from satellite imagery. Using Geographical Information System (GIS) software, the HUC2 regions were overlaid with the USDA CDL to identify the cropped areas (in acres) within each HUC2 region. In the original methods, five CDL years (2010-2014) were temporally aggregated, and the 111 crop categories native to CDL were grouped into 12 general classes: corn, cotton, soybean, wheat, grassland (*e.g*., pasture/hay), other crops (*e.g.*, clover, fallow field, sod/grass for seed), orchards and vineyards, other trees (*e.g*., managed/unmanaged forests), other grains (*e.g*., barley, buckwheat, canola, rye, sugarcane), other row crops (*e.g.*, peanuts, sugarbeet, sunflower, tobacco), vegetables and ground fruit, and Christmas tree orchards. The revised method was able to refine the orchard and vineyards category into the citrus, grapes, and other orchards (*e.g*., pome fruit, stone fruit, tree nuts) groups. Rice was also identified as a general crop; however, rice is modeled using a different surface water modeling approach (Pesticides in Flood Applications Model [PFAM]) (Young, 2013)[[7]](#footnote-7), separate from the PRZM5/VVWM ESA scenarios described here (**Section 3.1.3.1**).

The NASS Census of Agriculture[[8]](#footnote-8) (CoA) data were used to confirm growing regions for each crop group. If any crops were identified in the (CoA) that were not otherwise identified within a HUC2 region based on the CDL data, an input scenario was assigned for the corresponding HUC2 region-crop group combination. The results of this analysis are presented in **Table 2**. Cotton, citrus, and other trees were the only crop groups identified with no acreage within certain HUC2s. Based on this analysis, the HUC2 region-crop group combinations that had no acreage were excluded from the scenario selection process (identified in **Table 2** as blacked out cells). If a small acreage was noted for a HUC2 region-crop group combination, or the data were censored, a representative or surrogate scenario was identified for the HUC2.

Ten nonagricultural uses were also identified for modeling, including: mosquito adulticide, developed commercial areas, developed open space (*e.g*., recreational areas), golf, impervious, unspecified land cover (*e.g.*, nurseries), rangeland, residential, right-of-way, and wide area use (WAU).

The ESA aquatic modeling scenario files are named using the following convention: *crop\_group\_name*ESA*HUC2*. For instance, the corn scenario for HUC2 Region 1 has been named CornESA1.scn. If multiple meteorological stations were identified for a HUC2 region (**Section 3.1.3.1.4**), an “a” or “b” was added to the scenario name.

Table 2. Crop acres, by Crop Data Layer category and HUC2 region

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC 2****Region** | **Citrus** | **Corn** | **Soybean** | **Cotton** | **Grapes** | **Pasture/****Hay** | **Other Crops** | **Other Orchards** | **Other Trees** | **Other Grains** | **Other Row Crops** | **Wheat** | **Vegetables and Ground Fruit** | **Rice** |
| **01** | 0 | 358,677 | 13,619 | 0 | 2,021 | 2,299,065 | 114,592 | 62,791 | 4,170 | 184,053 | 8,590 | 7,021 | 396,400 | 0 |
| **02** | 0\* | 6,437,058 | 5,036,696 | 59,213 | 38,688 | 12,092,087 | 1,532,024 | 273,614 | 6,1976 | 983,442 | 13,301 | 2,700,573 | 363,841 | 1 |
| **03** | 1,796,914 | 7,239,956 | 8,899,772 | 9,269,747 | 11,545 | 39,422,125 | 7,788,600 | 3,226,616 | 1,771 | 1,731,680 | 4,197,455 | 5,323,073 | 570,695 | 24,764 |
| **04** | 0\* | 17,907,319 | 14,059,702 | 0 | 119,857 | 16,329,022 | 1,227,385 | 655,242 | 62,905 | 1,046,561 | 704,904 | 5,904,198 | 1,864,904 | 1 |
| **05** | 0\* | 22,682,226 | 22,162,761 | 4,721 | 70,189 | 24,208,197 | 642,247 | 97,674 | 16,635 | 248,699 | 46,590 | 5,034,839 | 288,184 | 33 |
| **06** | 0\* | 1,193,082 | 1,242,864 | 524,068 | 1,965 | 7,185,211 | 63,565 | 26,056 | 9,043 | 21,400 | 9,358 | 620,656 | 32,212 | 1 |
| **07** | 0\* | 57,748,484 | 55,163,033 | 687 | 19,399 | 29,514,129 | 253,889 | 87,353 | 2,810 | 1,497,659 | 493,776 | 5,310,065 | 1,788,603 | 5,395 |
| **08** | 1,520 | 8,813,986 | 17,114,672 | 9,837,633 | 505 | 8,081,052 | 4,653,753 | 125,823 | 0 | 1,978,115 | 61,285 | 6,191,869 | 133,488 | 7,344,580 |
| **09** | 28 | 7,663,065 | 12,777,001 | 0 | 87 | 9,971,751 | 2,000,936 | 993 | 0 | 4,237,298 | 4,150,689 | 16,916,992 | 2,857,719 | 0 |
| **10** | 3 | 58,577,416 | 46,412,519 | 348 | 3,397 | 197,647,865 | 22,118,235 | 53,689 | 373 | 15,313,057 | 5,647,024 | 53,147,405 | 4,873,417 | 11 |
| **11** | 3 | 11,325,835 | 7,112,783 | 3,419,503 | 2,714 | 84,690,678 | 10,819,174 | 793,640 | 7 | 11,297,418 | 329,644 | 29,754,728 | 127,076 | 889,374 |
| **12** | 62,610 | 4,227,885 | 449,521 | 11,049,544 | 5,414 | 45,393,414 | 5,585,025 | 919,877 | 460 | 7,034,185 | 319,888 | 7,812,459 | 96,428 | 795,211 |
| **13** | 62,383 | 159,088 | 3,287 | 329,819 | 4,018 | 20,898,576 | 1,070,392 | 346,844 | 336 | 508,002 | 19,513 | 261,299 | 241,345 | 4 |
| **14** | 0 | 206,984 | 373 | 61 | 2,297 | 11,069,850 | 399,163 | 49,552 | 0 | 198,050 | 18,472 | 358,150 | 175,998 | 0 |
| **15** | 89,642 | 211,196 | 8 | 1,001,954 | 27,830 | 7,409,916 | 2,087,833 | 217,939 | 0 | 381,894 | 500 | 433,978 | 244,647 | 0 |
| **16** | 13,251 | 314,238 | 45 | 0 | 5,884 | 9,306,753 | 1,011,709 | 65,559 | 127 | 588,750 | 746 | 919,038 | 80,873 | 0 |
| **17** | 5 | 1,990,248 | 5,601 | 0 | 184,405 | 36,862,718 | 6,271,300 | 1,423,664 | 109,830 | 2,680,486 | 823,676 | 11,674,036 | 4,203,380 | 0 |
| **18** | 949,461 | 2,634,163 | 68 | 2,656,390 | 1,882,619 | 26,638,360 | 4,184,381 | 9,985,246 | 0 | 1,958,542 | 357,380 | 2,708,716 | 1,829,285 | 1,161,516 |
| **19** | 0 | 0 | 0 | 0 | 0 | 42,437 | 0 | 15 | 0 | 5,348 |  | 182 | 299 | 0 |
| **20** | 2,440 | 8,374 | 0\* | 0 | 0\* | 3,456 | 504 | 117,527 | 0\* | 7 | 54 | 0\* | 13,983 | 0 |
| **21** | 20 | 1,026 | 0 | 0 | 0 | 0 | 0 | 48,987 | 0\* | 0 | 0 | 0 | 2,869 | 0 |

\* Although CDL data do not indicate the crop is grown in this HUC2, NASS data indicate small amounts of the crop is grown, so scenarios are developed to facilitate exposure modeling of these minor crops.

##### Development of Representative Scenarios

The standard input scenarios were binned based on location and crop into HUC2 region-crop group combinations. The scenario with the highest runoff curve number was identified per HUC2 region-crop group combination, as it represented the highest runoff potential. For those HUC2 region-crop group combinations where input scenarios were not available, a surrogate scenario (with the highest runoff potential) from a neighboring HUC2 region was selected.

**Table 3** identifies the surrogate scenarios used for ESA aquatic exposure modeling. For nonagricultural uses of adulticide, developed, right-of-way (ROW), and wide area use, the CArightofwayRLF\_V2 scenario was used. For impervious and residential uses, the CAImperviousRLF and CAresidentialRLF scenarios were used, respectively.

##### Spatial Delineation, Weather Data

Currently, each of the existing scenarios are linked to a specific weather station location from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center’s (NCDC) Solar and Meteorological Surface Observation Network (SAMSON). The SAMSON dataset[[9]](#footnote-9) provides the daily rainfall, pan evaporation, solar radiation, temperature, and wind speed for 242 National Weather Service (NWS) locations, spanning the years 1961 to 1990. For each of the ESA scenarios, a representative SAMSON weather station was assigned from among the stations located within the corresponding HUC2 region, based on the highest 30-year rainfall level (**Table 4**).

In order to identify the representative station for use with the ESA scenarios, the 242 meteorological stations were grouped by HUC 2 and a cumulative 30-year precipitation value was estimated. The meteorological station with the median cumulative precipitation value for a HUC 2 region is selected as the representative weather station except where there was not a large difference in the precipitation values (*i.e*., the maximum cumulative 30-year precipitation value for a HUC2 was three times greater than the minimum value). For HUC2 regions where a large rainfall difference occurred, the median precipitation value was used as a demarcation between high-precipitation and low-precipitation groups. The median station for both the high-precipitation and the low-precipitation groups were identified as representative weather stations and two sets of modeling were conducted for each HUC2 region (see **Section 3.1.3.1.2** for how these weather stations are identified within the scenarios). For HUC2 regions 15, 16 and 20, a large disparity existed between the highest precipitation station and the remaining stations in the HUC2. For these HUC2 regions, the highest precipitation weather station was selected along with the weather station with the median cumulative 30-year precipitation value for the remaining stations.

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##### Scenario Modifications for Evaluating Wetlands

For modeling exposures to listed species in wetlands, the same scenarios discussed above were modified to account for a 15 cm benthic depth to represent the typical active root zone for wetland species of forbs and woody plants. These scenarios are only used when modeling wetlands (bin 1). Please refer to the PAT User Guide for more details.

Table 3. PRZM5/VVWM surrogate scenarios used for ESA aquatic modeling

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC 2** | **Corn** | **Soybean** | **Citrus** | **Cotton** | **Developed, Open Space/****Golf** | **Grassland/Rangeland/****Other Crops** | **Grapes** |
| **01** | MIbeansSTD | MIbeansSTD |  |  | PATurfSTD | ILalfalfaNMC | NYGrapesSTD |
| **02** | PAcornSTD | PAcornSTD | FLcitrusSTD | NCcottonSTD | PATurfSTD | PAturfSTD | NYGrapesSTD |
| **03** | NCcornWOP | NCcornWOP | FLcitrusSTD | MSCottonSTD | FLTurfSTD | NCalfalfaOP | NYGrapesSTD |
| **04** | MIbeansSTD | MIbeansSTD | FLcitrusSTD |  | PATurfSTD | PAturfSTD | NYGrapesSTD |
| **05** | OHCornSTD | OHCornSTD | FLcitrusSTD | MSCottonSTD | PATurfSTD | ILalfalfaNMC | NYGrapesSTD |
| **06** | NCcornWOP | NCcornWOP | FLcitrusSTD | MSCottonSTD | FLTurfSTD | NCalfalfaOP | NYGrapesSTD |
| **07** | ILcornSTD | ILcornSTD | FLcitrusSTD | MSCottonSTD | PATurfSTD | ILalfalfaNMC | NYGrapesSTD |
| **08** | MSCornSTD | MSCornSTD | FLcitrusSTD | MSCottonSTD | FLTurfSTD | TXalfalfaOP | NYGrapesSTD |
| **09** | NDCornOP | NDCornOP | FLcitrusSTD |  | PATurfSTD | MNalfalfaOP | NYGrapesSTD |
| **10** | KScorn | KScorn | FLcitrusSTD | STXcottonNMC | PATurfSTD | ILalfalfaNMC | NYGrapesSTD |
| **11** | NECornSTD | NECornSTD | FLcitrusSTD | STXcottonNMC | FLTurfSTD | TXalfalfaOP | NYGrapesSTD |
| **12** | STXcornNMC | STXcornNMC | STXgrapefruitNMC | STXcottonNMC | FLTurfSTD | TXalfalfaOP | NYGrapesSTD |
| **13** | TXcornOP | TXcornOP | FLcitrusSTD | STXcottonNMC | CATurfRLF | TXalfalfaOP | NYGrapesSTD |
| **14** | TXcornOP | TXcornOP |  | STXcottonNMC | CATurfRLF | TXalfalfaOP | NYGrapesSTD |
| **15** | TXcornOP | TXcornOP | CAcitrus\_WirrigSTD | CAcotton\_WirrigSTD | CATurfRLF | TXalfalfaOP | CAgrapes\_WirrigSTD |
| **16** | TXcornOP | TXcornOP | CAcitrus\_WirrigSTD |  | CATurfRLF | TXalfalfaOP | CAgrapes\_WirrigSTD |
| **17** | ORswcornOP | ORswcornOP | CAcitrus\_WirrigSTD |  | CATurfRLF | ORwheatOP | ORHopsSTD |
| **18** | CAcornOP | CAcornOP | CAcitrus\_WirrigSTD | CAcotton\_WirrigSTD | CATurfRLF | CArangelandhayRLF\_V2 | CAgrapes\_WirrigSTD |
| **19** |  |  |  |  | CATurfRLF | ORwheatOP |  |
| **20** | FLcorn | NCcornWOP | FLcitrusSTD |  | CATurfRLF | FLTurf | CAgrapes\_WirrigSTD |
| **21** | FLcorn |  | FLcitrusSTD |  | FLTurfSTD |  |  |

**Table 3. PRZM5/VVWM surrogate scenarios used for ESA aquatic modeling (continued)**

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC 2** | **Non-specified land cover** | **Other Orchards** | **Other Trees /****Xmas Tree1** | **Other Grain** | **Other Row Crop** | **Wheat** | **Vegetables/****Ground Fruit** |
| **01** | MInurserySTD | PAapplesSTD\_V2 | NYgrapesSTD | PAalfalfaOP | MEpotatoSTD | PAalfalfaOP | MEpotatoSTD |
| **02** | NJnurserySTD\_V2 | PAapplesSTD\_V2 | PAapplesSTD\_V2 | PAalfalfaOP | NJmelonSTD | PAalfalfaOP | PAvegetableNMC |
| **03** | FLnurserySTD\_V2 | NCappleSTD | FLcitusSTD | FLsugarcaneSTD | NCpeanutSTD | NCalfalfaOP | FLpotatoNMC |
| **04** | MInurserySTD |  MICherriesSTD | MIcherriesSTD |  ILalfalfaNMC | MImelonsSTD | NDwheatSTD | MImelonsSTD |
| **05** | NJnurserySTD\_V2 | PAapplesSTD\_V2 | PAapplesSTD\_V2 | KSsorghumSTD | NCpeanutSTD | KSsorghumSTD | MIbeansSTD |
| **06** | TNnurserySTD\_v2 | NCappleSTD | NCappleSTD | NCalfalfaOP | NCcornWOP | NCalfalfaOP | FLpotatoNMC |
| **07** | TNnurserySTD\_v2 |  MICherriesSTD | FLcitusSTD | ILalfalfaNMC | ILcornSTD | ILalfalfaNMC | ILbeansNMC |
| **08** | FLnurserySTD\_V2 | NCappleSTD | FLcitusSTD | LAsurgarcaneSTD  | MOmelonSTD | ILalfalfaNMC | MOmelonSTD |
| **09** | MInurserySTD |  MICherriesSTD | MIcherriesSTD | NDcanolaSTD | Mnsugarbeet | NDwheatSTD | MNsugarbeatSTD  |
| **10** | TNnurserySTD\_v2 | ORfilbertsSTD | FLcitusSTD | KSsorghumSTD | KScorn | NDwheatSTD |  MNsugarbeatSTD  |
| **11** | TNnurserySTD\_v2 | OrchardBSS | FLcitusSTD | TXwheatOP | NECornSTD | TXwheatOP | STXmelonNMC  |
| **12** | NurseryBSS\_V2 | OrchardBSS | OrchardBSS |  TXwheatOP | STXcornNMC  | ILalfalfaNMC | STXmelonNMC |
| **13** | NurseryBSS\_V2 | OrchardBSS | OrchardBSS | TXwheatOP | STXcornNMC  | ILalfalfaNMC | STXmelonNMC |
| **14** | TNnurserySTD\_v2 | OrchardBSS | OrchardBSS | TXwheatOP | STXcornNMC  | ILalfalfaNMC | STXmelonNMC |
| **15** | NurseryBSS\_V2 | CAalmond\_WirrigSTD | CAcitrus\_WirrigSTD | TXwheatOP | STXcornNMC  | TXwheatOP | CALettuceSTD |
| **16** | CAnurserySTDV | CAalmond\_WirrigSTD | CAcitrus\_WirrigSTD | TXwheatOP | STXcornNMC  | ILalfalfaNMC | STXmelonNMC |
| **17** | ORnursery | ORappleSTD | ORxmastresSTD | ORwheatOP | ORhopsSTD | ORwheatOP | ORsnbeanSTD |
| **18** | CAnurserySTDV | CAalmond\_WirrigSTD | CAalmond\_WirrigSTD |  CAWheatRLF\_V2 | CArowcropRLF\_V2 | CAWheatRLF\_V2 | CAlettuceSTD  |
| **19** | ORnursery | ORappleSTD | ORxmastresSTD | ORwheatOP |  | ORwheatOP | ORsnbeanSTD |
| **20** | FLnurserySTD\_V2 | CAalmond\_WirrigSTD | FLcitrusSTD | FLsugarcaneSTD | FLpotatoNMC |  | FLtomatoSTD |
| **21** | FLnurserySTD\_V2 | PRCoffeeSTD | PRCoffeeSTD |  |  |  | FLtomatoSTD |

1. Christmas tree scenario only developed for HUC2 regions 1-19.

Table 4. Representative weather stations by HUC2 region 1

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC2** | **Value** | **WBAN** | **Precip (cm)** | **Precip Range (cm)** | **HUC2** | **Value** | **WBAN** | **Precip (cm)** | **Precip Range (cm)** |
| 1 | Median | 14740 | 3367 | 2774 – 3641 | 13 | Median | 23044 | 673 | 581 – 578 |
| 2 | Median | 13733 | 3120 | 2759 – 3629 | 14 | Median | 24027 | 760 | 661 – 856 |
| 3 | Median | 13874 | 3870 | 3019 – 5009 | 15 | Median (1) | 03103 | 585 | 315 – 915 |
| 4 | Median | 14839 | 2514 | 1890 – 3183 | 15 | Highest (2) | 23183 | 1740 | 1740 |
| 5 | Median | 93814 | 3151 | 2650 – 3607 | 16 | Median (1) | 24127 | 628 | 475 – 877 |
| 6 | Median | 13891 | 3594 | 3083 – 4362 | 16 | Highest (2) | 24128 | 1238 | 1238 |
| 7 | Median | 14933 | 2525 | 2091 – 2979 | 17 | Median (1) | 24156 | 928 | 608 – 1438 |
| 8 | Median | 13964 | 4641 | 3992 – 4746 | 17 | Median (2) | 24221 | 3762 | 1438 – 6291 |
| 9 | Median | 14914 | 1487 | 1342 – 1859 | 18 | Median (1) | 23232 | 756 | 296 – 909 |
| 10 | Median (1) | 14935 | 1107 | 838 – 1390 | 18 | Median (2) | 23188 | 1338 | 909 – 2862 |
| 10 | Median (2) | 24029 | 1902 | 1390 – 3282 | 19 | Median (1) | 26415 | 913 | 347-1215 |
| 11 | Median (1) | 13963 | 1491 | 710 – 2220 | 19 | Median (2) | 26528 | 2224 | 1215-11525 |
| 11 | Median (2) | 23047 | 3121 | 2220 – 3875 | 20 | Median (1) | 22521 | 1682 | 1598 – 3287 |
| 12 | Median (1) | 03927 | 1861 | 1141 – 2397 | 20 | Highest (2) | 21504 | 9891 | 9891 |
| 12 | Median (2) | 13897 | 2569 | 2397 – 4359 | 21 | Median | 11641 | 3974 | 3974 |

1 WBAN - Weather Bureau Army Navy. The number in parenthesis indicates the median station for the low-precipitation group (1) and the median or highest station for the high-precipitation group (2).

#### Aquatic Habitat Bins

In response to the National Academy of Sciences (NAS) Report[[10]](#footnote-10) recommendations, the National Marine Fisheries Service and the Fish and Wildlife Service developed 10 habitat bins (including nine aquatic bins and an aquatic associated terrestrial habitat) for use in the ESA pesticide exposure assessments (**Table 1**). The nine aquatic habitat bins (or water body types) are used in aquatic exposure modeling and are intended to represent the range of potential habitats where exposure to aquatic endangered species and designated critical habitat may occur. The bins are linked to the aquatic modeling scenarios (**Section 3.1.3.1**) and specific species to provide spatially and temporally-relevant estimated exposure concentrations (EEC) for each habitat. The nine aquatic habitat bins are used in the BEs for both Step 1 and Step 2 and will be used for the Biological Opinions in Step 3.

Each habitat bin was developed to represent three general categories: freshwater static waters, freshwater flowing waters, and estuarine/marine waters. For each bin, representative dimensions and flow regimes were defined (**Table 1**). The rationale used to develop each habitat bin is described below.

**Terrestrial.** Terrestrial habitats include both upland and aquatic-associated habitats, as well as terrestrial areas adjacent to treated areas that may receive runoff and spray drift and expose listed plants or species that rely on plants for prey, pollination, habitat, or dispersal (PPHD). These terrestrial plant exposure zones (T-PEZs) have a length of 316 m (equal to the length of the edge of the treated field), and a width of 30 m. The width of the T-PEZ represents the distance that overland surface flow can travel before sheet flow transitions to concentrated flow. Based on an evaluation of maximum root depths across different crops, the depth of the T-PEZ is 15 cm.

Aquatic-associated terrestrial habitats are addressed below.

**Bin 1.** *Aquatic-associated terrestrial habitats*. A number of endangered species utilize habitats that include both aquatic and terrestrial characteristics such as riparian zones, beaches, intertidal zones, intermittent streams, and seasonal wetlands. While estimation of surface water concentrations is relevant to species that utilize these habitats, alternative routes and pathways of exposure are also relevant and need to be estimated for species that occupy them during periods when they are not inundated with water.

For listed plant species and listed species that rely on plants for PPHD, bin 1 is being modeled as 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 3.3**).

**Flowing water habitats.**  Flowing water habitats vary considerably in depth, width, and velocity, which influence both initial concentration as well as rates of dissipation of pollutants. At least three reference points seemed reasonable to estimate concentration ranges in flowing water habitats. Distinctions were made based on the range in flow rates. Flow rates accounted for the velocity of the water (influenced by the gradient and other factors) as well as width and depth of the habitat. A higher velocity stream may have an equivalent flow rate to a stream with lower velocity and greater cross-sectional area (width x depth). In the former case, higher initial concentrations would be expected because the habitat has lower volume. However, faster dissipation would also be expected given the pesticide will be moved off-site quickly considering the greater velocity. Flow rates vary temporally and spatially in these habitats and are influenced by a number of factors. For example, bends in the shoreline, shoreline roughness, and organic debris can create back currents or eddies that can concentrate allochthonous inputs. Dams and other water control structures also significantly influence flow. Some small streams and channels are intermittent and can become static and temporally cut off from connections with surface water flows. Low flow habitats may also occur on the margins of higher flow systems (*e.g.*, floodplain habitats associated with higher flowing rivers) or in caves or other sub-surface environments.

**Bin 2**. *Low-flow habitats* (0.001 – 1 m3/sec). Some examples of low flow habitats include springs, seeps, brooks, small streams, floodplain habitats (oxbows, side channels, alcoves, *etc*.), dendritic channels that occur within exposed intertidal areas, and distributary channels in estuaries on the incoming and slack tides. Model input parameters for the “low-flow habitat” were selected to represent the lower end of the flow rate range and habitat dimensions are consistent with previous modeling to estimate exposure in habitats used by listed salmonids. The formula in the attached EPA link was used to estimate the speed for each flowing water habitat assuming the muddy substrate coefficient of 0.8 (<http://water.epa.gov/type/rsl/monitoring/vms51.cfm>). Considering the dimensions of this habitat and the flow rate, the velocity of the water in this system would be moving downstream very slowly (about 1 foot/min).

**Bin 3**. *Moderate-flow habitats* (1 - 100 m3/sec). This range in flow rates is comparable to that found in small to large streams (~35 -3,500 cfs). It may also be representative of smaller rivers and habitats along the margins of larger rivers systems where depths and flow rates within the thalweg (i.e., the middle of the chief navigable channel of a waterway) can be substantially greater. Model input parameters were chosen to represent habitats at the lower flow volume end of this range. The estimated velocity of this system is approximately 0.14 m/s, or about 0.3 mph.

**Bin 4**. *High-flow habitats* (>100 m3/sec). Water bodies characterized as rivers typically have flow rates of >100 m3 (~3,500 cfs), and very large rivers may exceed a mile in width and have flow volumes that exceed 10,000 m3. Model input parameters were chosen to represent habitats at the lower flow volume end of this range. This represents a faster moving system with an estimated velocity of about 1.4 m/s (3+ mph).

**Static aquatic habitats.**  Pools, ponds, lakes and several other aquatic habitats are relatively static. Flow is less likely to substantially influence exposure in these habitats because it is generally lacking. Static habitats are broken up into three size categories below based on dilution volume.

**Bin 5**. *Low-volume static* (0 – 100 m3). Some examples of low volume habitats used by endangered species include vernal pools, small ponds, floodplain habitats that are cut off from main channel flows, underground pools, and seasonal wetlands. Model parameters were selected to represent the lower end of the range used by listed species.

**Bin 6**. *Moderate-volume static* (100 – 20,000 m3). Some examples of habitats in this category include ponds, some wetlands, and even small shallow lakes.

**Bin 7**. *High-volume static* (>20,000 m3). This volume was chosen as a point of reference because it is equivalent to the size of habitat typically modeled by EPA in PRZM5/VVWM, a one-hectare (~2.5 acre) water body that is 2 meters deep. Additional categories could be added to address endangered species that occur in larger volume habitats; however, it may not be necessary to model larger habitats for step 2. The smaller volume habitats specified here could be used to estimate concentrations around the outer margins of larger lakes.

**Estuarine and marine habitats.** Three marine habitats are identified and characterized by their position relative to the shoreline and each other. Current pesticide models do not account for transport via tidal and wind generated currents in marine systems. As such, surrogate bins have been identified among the flowing and static bins to represent pesticide concentrations that may be expected in these environments (see **Section 3.1.3.3.3** below for more information on which surrogate bins are selected).

**Bin 8**. *Intertidal nearshore*. The intertidal zone represents the nearshore area between the ordinary high-water mark and the extreme low water mark. Depth of intertidal habitats are variable and generally range from 0 to <10 m. Depth within the intertidal habitat depends on the tidal cycle and tidal range. At some locations along the shoreline, there is no discernable difference between high and low tides (tidal range). The greatest tidal range is about 16.3 meters at the Bay of Fundy in eastern Canada. The width of the intertidal area is also location specific and depends on the tidal range and the gradient/slope of the substrate. A depth of 0.5 m and width of 50 m were selected to represent this habitat, considering that exposure to the more vulnerable microhabitats could occur within the intertidal zone.

**Bin 9**. *Subtidal nearshore*. The subtidal nearshore zone represents the area between the intertidal zone and the continental shelf. The range in depth extends from 0 meters where it meets the intertidal zone to approximately 200 meters near the continental shelf. To estimate concentrations near the intertidal interface of this habitat, a depth of 5 meters and a width of 200 meters were selected.

**Bin 10**. *Offshore marine*. Offshore marine habitats are generally >200 meters in depth and cover vast areas. A habitat definition of 200m deep and 300 m wide (the approximate limit for AgDRIFT model) is suggested.

#### Watershed Sizes

To incorporate the aquatic habitat bins into the surface water modeling framework, an appropriate watershed size is needed for each water body type. For example, when conducting FIFRA ecological risk assessments, EPA uses the Standard Pond, which has a 10-hectare watershed draining to a 1-hectare pond. This watershed area to water body area ratio (10:1) maximizes the amount of runoff received by the water body yet minimizes runoff events that exceed the Standard Pond volume (*i.e*., 20,000 m3). Previous BEs utilized a GIS analysis by HUC2 to develop watershed sizes for flowing waterbodies and a PWC analysis designed to ensure runoff volumes do not excessively overflow the volumes of static waterbodies. Unfortunately, modeling using the watershed sizes in the pilot BEs resulted in excessive overprediction of pesticide concentrations (*i.e*., modeled concentrations several orders of magnitude higher than observed concentrations) in flowing waterbodies. In static waterbodies, as the ratio of the watershed area to waterbody volume was similar for medium and large volume waterbodies, pesticide concentrations in these two waterbodies tended to be the same.

In an effort to streamline the modeling and generate protective, realistic EECs, EPA relied on two standard waterbodies which have been traditionally used in EFED’s FIFRA assessments to estimate EECs for the various bins. The standard farm pond (a 10-hectare watershed draining into a 20,000 m3 waterbody) was used to develop EECs for the medium and large static bins (*e.g.*, bins 6 and 7). As mentioned above, the EECs generated in the previous BEs for the medium and large static bins were generally the same, so streamlining the modeling for these two waterbodies seemed appropriate. To assess EECs in the medium and large flowing bins (*e.g*., bins 3 and 4), EPA used its flowing waterbody conceptual model, the index reservoir (a 172.8-hectare watershed draining into roughly a 144,000 m3 waterbody). While EPA acknowledges the flowrates for the medium and large flowing bins are much higher than those generated for the index reservoir, EPA believes the index reservoir is an SAP-approved conceptual model that will generate protective EECs for the medium/large flowing bins. For the smallest flowing and static bins (bin 2 and 5), which represent headwater such as springs, seeps, and floodplain areas, EPA derived edge of field (treated area) estimates, using the PWC edge of field calculator tool, from the PRZM5 daily runoff file (*e.g*., ZTS file).

Aquatic EECs resulting from spray drift only are estimated using equations derived from AgDRIFT and the original waterbody dimensions for the 6 aquatic bins (**Table 1**).

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####  Application Date Selection

In selecting application dates for aquatic modeling, several factors are considered including: label directions; timing of pest pressure; weather conditions; and pre-harvest restriction intervals. Agronomic information is used to determine the timing of pest pressure and seasons for different crops. General sources of information include crop profiles[[11]](#footnote-11), agricultural extension bulletins, and available state-specific use information.

Weather information is also considered, as pesticide loading to surface water is affected by precipitation events. Model simulations evaluate application(s) during the wettest month (*e.g*., the month with the highest daily average precipitation), provided label information indicates that it can be used during this timeframe. For instance, if a pesticide is applied postemergence and the wettest month occurs between emergence and harvest, the wettest month is used. However, if the wettest month occurs before emergence, then the next wettest month that meets the criteria (*e.g*., occurs between emergence and harvest) is used. A random application date (*e.g*., the first of the month, the middle of the month) is selected in an effort to maintain the probability of the distribution of environmental exposure concentrations generated. A listing of the months by decreasing average daily precipitation is provided in **Table 5**.

Table 5.Weather analysis to determine wettest months in each HUC2 region

| **HUC2** | **Wettest Month** |
| --- | --- |
| **1st** | **2nd** | **3rd** | **4th** | **5th** | **6th** | **7th** |
| 1 | 11 | 5 | 4 | 9 | 12 | 6 | 8 |
| 2 | 7 | 5 | 10 | 8 | 6 | 3 | 2 |
| 3 | 3 | 2 | 7 | 1 | 4 | 12 | 5 |
| 4 | 4 | 8 | 9 | 7 | 6 | 5 | 3 |
| 5 | 5 | 3 | 7 | 6 | 4 | 11 | 8 |
| 6 | 3 | 7 | 12 | 2 | 1 | 5 | 6 |
| 7 | 6 | 8 | 7 | 5 | 9 | 4 | 10 |
| 8 | 7 | 2 | 8 | 4 | 12 | 9 | 1 |
| 9 | 6 | 7 | 5 | 8 | 9 | 4 | 10 |
| 10a | 6 | 5 | 9 | 7 | 8 | 4 | 3 |
| 10b | 5 | 6 | 4 | 9 | 10 | 3 | 7 |
| 11a | 5 | 11 | 4 | 3 | 10 | 6 | 9 |
| 11b | 6 | 8 | 7 | 5 | 9 | 10 | 4 |
| 12a | 5 | 4 | 10 | 9 | 6 | 3 | 2 |
| 12b | 9 | 5 | 6 | 8 | 10 | 7 | 4 |
| 13 | 9 | 8 | 7 | 10 | 6 | 12 | 2 |
| 14 | 5 | 4 | 9 | 6 | 7 | 3 | 8 |
| 15a | 7 | 8 | 3 | 12 | 2 | 9 | 1 |
| 15b | 12 | 8 | 9 | 3 | 7 | 2 | 11 |
| 16a | 4 | 3 | 5 | 10 | 12 | 2 | 11 |
| 16b | 11 | 6 | 12 | 4 | 5 | 3 | 1 |
| 17a | 12 | 11 | 1 | 2 | 3 | 10 | 4 |
| 17b | 5 | 3 | 4 | 11 | 12 | 6 | 1 |
| 18a | 1 | 2 | 11 | 3 | 12 | 4 | 10 |
| 18b | 1 | 3 | 2 | 12 | 11 | 4 | 10 |
| 19a | 7 | 6 | 8 | 9 | 5 | 10 | 11 |
| 19b | 8 | 9 | 7 | 10 | 6 | 12 | 11 |
| 20a | 4 | 11 | 3 | 12 | 2 | 5 | 1 |
| 20b | 12 | 1 | 11 | 2 | 10 | 3 | 4 |
| 21 | 11 | 5 | 10 | 9 | 8 | 12 | 7 |

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		2.
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			2.
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			4.

#### Spray Drift Exposure

AgDRIFT v 2.1.1 (Spray Drift Task Force, 2011)[[12]](#footnote-12) is used to evaluate the deposition fractions for aerial, ground, and orchard applications based on label specifications. These fractions are then used to estimate the fraction of pesticide applied that reaches the water body by spray drift. If spray drift buffer zones are specified on the label, the distance is included in the AgDRIFT analysis. If spray drift buffer zones are not specified on the label, then the water body is assumed to adjoin the treated field. Default spray drift deposition values for the various water bodies, when the assessor is considering spray drift only, are provided in **Table 6**. If more refined analysis is required, chemical-specific values will be derived and incorporated into a spray drift appendix. For modeling using PRZM5/VVWM, EPA uses drift estimates derived using the standard farm pond and index reservoir waterbody dimensions.

Table 6. Default spray drift fractions for use in PRZM5/VVWM

|  |  |  |
| --- | --- | --- |
| **Application Method** | **Drop size Distribution / Category** | **Spray drift fraction (unitless)1** |
| **Bin 1** | **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| Aerial | Very fine to fine | 0.242 | 0.472 | 0.414 | 0.291 | 0.486 | 0.401 | 0.195 |
| **Fine to medium (default)** | 0.125 | 0.437 | 0.320 | 0.167 | 0.469 | 0.297 | 0.093 |
| Medium to coarse | 0.089 | 0.424 | 0.284 | 0.123 | 0.462 | 0.257 | 0.063 |
| Coarse to very coarse | 0.068 | 0.412 | 0.261 | 0.097 | 0.456 | 0.233 | 0.047 |
| Ground high boom | **Very fine to fine (default)** | 0.062 | 0.620 | 0.294 | 0.089 | 0.778 | 0.252 | 0.042 |
| Fine to medium/coarse | 0.017 | 0.215 | 0.079 | 0.024 | 0.336 | 0.067 | 0.012 |
| Ground low boom | Very fine to fine | 0.027 | 0.365 | 0.140 | 0.039 | 0.528 | 0.118 | 0.019 |
| Fine to medium/coarse | 0.011 | 0.154 | 0.054 | 0.016 | 0.251 | 0.045 | 0.008 |
| Airblast | **Sparse (default)** | 0.042 | 0.372 | 0.219 | 0.064 | 0.418 | 0.192 | 0.027 |
| Normal | 0.001 | 0.007 | 0.004 | 0.002 | 0.008 | 0.004 | 0.001 |
| Dense | 0.015 | 0.094 | 0.060 | 0.021 | 0.104 | 0.054 | 0.010 |
| Vineyard | 0.002 | 0.025 | 0.013 | 0.004 | 0.030 | 0.011 | 0.002 |
| Orchard | 0.022 | 0.174 | 0.104 | 0.033 | 0.195 | 0.092 | 0.015 |

1 Estimated using Tier 1 in AgDRIFT 2.1.1 and the following water body widths: Bin 1 – 64 m, Bin 2 – 2 m, Bin 3 – 8 m, Bin 4 – 40 m, Bin 5 – 1 m, Bin 6 – 10 m, and Bin 7 – 100 m.

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## Pesticide Flooded Application Model (PFAM)

PFAM is a model 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, and allows for more flexibility in refinement of assessments when needed. PFAM allows for the simulation of pesticide applications to a dry field and degradation in soil before water is introduced to the field.

PFAM is used in the ESA modeling effort to estimate pesticide concentrations in flood water releases from a paddy or bog. The concentrations are representative of the water releases from the field and not mixed with any additional water (*i.e*., receiving water body). As a result, the estimated concentrations presented may be greater than those expected in adjacent water bodies due to additional degradation and dilution. While PFAM can additionally simulate concentrations in a receiving water body, a validated conceptual model has not yet been developed for ecological risk assessment purposes, so this feature is not included here.

Differences in the concentration of the pesticide in the flood water compared to an adjacent water body depend on 1) the length of time the pesticide is in the flooded field, 2) the distance the water travels between the flooded field and the receiving water body, 3) the amount of dilution in the receiving water body, and 4) whether the flood water is mixed with additional water that also contains the pesticide.

## Plant Assessment Tool (PAT)

For modeling exposures to listed plants and species that rely on plants for PPHD located near treated fields, EPA is using the Plant Assessment Tool (PAT). 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 listed species, runoff and erosion are modeled using PRZM5 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 PRZM5/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 PRZM5/VVWM models and the standard farm pond. For more information on PAT, please refer to the PAT User Manual.

## Probabilistic PRZM5/VVWM Modeling for Use in the MAGtool

EPA attempted to incorporate probabilistic methods into its new analysis to determine the likelihood of exposure and effects to an individual of a listed species. The goal of the probabilistic analysis was to more fully capture and characterize variability in the range of potential risks that could occur based on the inherent variability in the most influential input parameters used in EPA’s models. The MAGtool is a new tool that combines toxicological information, exposure analysis and spatial analysis into one place. For each species range and critical habitat, the MAGtool provides the number of individuals predicted to be impacted under the assumptions of the analysis.

In the aquatic environment, exposure concentrations were drawn from predicted EECs within a relevant size water body for a species. Different distributions of maximum annual daily EECs, designed to represent the variability in EECs from year to year, were considered, depending on if the species was located in flowing or static waterbodies. For the static and low-flow waterbodies, the distribution of maximum daily EECs from the 30 years of data was used based on the assumption that a species will not leave that static bin and could be exposed to the maximum exposure concentrations for a given year. For medium and high-flow flowing waterbodies, there will be movement of the species, as well as the water, within the water bodies and there is higher variability and uncertainty in the expected exposure concentrations. In this case, the distribution of daily EECs based on the 90-day window around the maximum annual daily concentration was used in the analysis. For each PRZM5/VVWM run, and each year of the run, the 90 highest daily concentrations of the pesticide around the annual maximum daily concentration were summarized and used in the MAGtool to provide a means to evaluate the impact of the variation in EECs to which the species may be exposed.

Other factors can impact the concentrations in a water body under varied application times, rates and conditions. To try and capture some of this variability, the influence of 2 additional factors, application date and hydrologic soil group, were considered in the distribution of EECs. These factors were chosen as they can have a substantial impact on EECs and are expected to vary considerably in real world applications.

Simulations were conducted with PRZM5/VVWM to determine the EECs for single applications at the maximum application rate using the date associated with the month with the maximum precipitation within a realistic application window for each scenario and bin (details on bins and modeling assumptions provided in the exposure chapter analysis section of pilot BEs). The same simulations were run using alternate application dates that would fall within a reasonable application window (generally April to August, or the relevant application window for the area). Factors were developed which related the EEC associated with the original chosen application date to the EEC from the randomly selected application date. For example, if the EEC from the original analysis based on a May 1 application date was 80 μg/L and the randomly selected date yielded an EEC of 70 μg/L, the factor applied would be 0.875 (70/80 = 0.875). A distribution of factors was created based on all the variable dates modeled.

A similar analysis was conducted using different hydrologic soil groups. Original PRZM5/VVWM modeling used ESA scenarios developed with hydrologic soil groups. The hydrologic soil groups used in the ESA scenarios were considered conservative and generated high levels of runoff. The analysis looked at hydrologic soil groups which would reduce the runoff from a use site, resulting in lower EECs. Again, PRZM5/VVWM modeling was done using the original aquatic runs done for the BEs, but were conducted using modified ESA scenarios designed to represent different hydrologic soil groups. Scaling factors (which were pesticide and scenario specific) were developed by comparing the EECs from the original PRZM5/VVWM runs to the modified runs.

# Use of Monitoring Data

Pesticide monitoring data are available for various media including water, sediment, air, precipitation and biota. These data are collected by federal, state, and local agencies, universities, registrants, and volunteers. Generally, these data are used to characterize exposure, identify trends over time, and assess mitigation measures. The occurrence of pesticides in environmental media depends on factors such as:

* Pesticide physical-chemical properties;
* Spatial pesticide use patterns, crop and management practices, soil and hydrologic vulnerabilities, and weather including rainfall, temperature, humidity and wind;
* Intensity and timing of pesticide applications and coincidence with the timing of the sampling and weather events;
* Year-to-year temporal patterns at any given location reflecting changes in cropping and pesticide use as well as variations in weather from year to year; and
* Extent of impervious surfaces in urban areas and hydrology of engineered urban stormwater systems.

Minimum elements (*i.e*., ancillary data) needed to evaluate monitoring data include:

* Study objective (*i.e.*, purpose and design of the monitoring study); a copy of a report describing the purpose and design of the monitoring study or internet web address leading to this information would be useful if available;
* Location description (latitude & longitude, if possible, or other reliable location information);
	+ Pesticide application sites
	+ Monitoring station/sample site (and distance from pesticide application site)
* Date(s) sampled;
* Sample media (*e.g.*, water, filtered water, bed sediment, biota)
	+ Water body type (stream, river or other flowing body; lake, reservoir, or other static body; groundwater; nature of aquifer, *e.g.,* surficial or confined; depth to groundwater and screen depth); and purpose (*e.g*., drinking water, irrigation, and monitoring);
	+ Water body parameters (width, depth, flow rate)
* Pesticide(s) analyzed and reported concentration;
* Analytical method and detection limit (LOD)/limit of quantitation (LOQ).

Other important information (*i.e*., ancillary data) that aids in evaluating and interpreting monitoring data include:

* Quality assurance (QA)/quality control (QC) for sample collection and analytical methods, including a discussion of any limitations of the data;
* Sample collection method [*e.g.*, single [point often called grab) or composite];
* Time of sample [*e.g.*, date, time; duration (if a composite)];
* Meteorological data (*e.g*., temperature, wind speed, wind direction, and rainfall);
* Soil and/sediment characteristics (*e.g.*, organic carbon, bulk density);
* Agronomic practices (*e.g.*, irrigation, land use, including cropping pattern, agriculture/urban);
* Pesticide usage [application date, rate, and method (including release height, droplet spectrum); and
* Sampled media characteristics (*e.g*., organic carbon, bulk density, pH, hardness, turbidity)

This section is intended to describe the process by which monitoring data are evaluated and used in ecological exposure assessments, including Endangered Species Assessments. This discussion is divided into four areas: 1) evaluation of monitoring data, 2) the use of monitoring data for quantitative purposes (*i.e*., reasonable upper bound exposure concentration), 3) the use of monitoring data to evaluate downstream transport of pesticides, and 4) the qualitative use of monitoring data (*e.g*., for characterization as a line of evidence in a weight-of-evidence approach).

## Evaluation of Monitoring Data

Monitoring data provide snapshots of pesticide concentrations in time at specific locations under the conditions which the data are collected. Supporting information or ancillary data are critical to understanding the monitoring data in context of overall pesticide exposures in the environment.

Monitoring data where 1) sampling occurs in a high use area, 2) sampling occurs during the time frame in which pesticides are expected to be used, and 3) the sampling is frequent enough to estimate exposures for the endpoints of concern, are more informative to risk assessment, as compared to, monitoring data where these factors are unknown or did not occur. Evaluation of the monitoring data is critical because it provides context to model estimated pesticide exposure concentrations and helps risk assessors better understand exposure on a regional or local scale under actual use conditions.

Monitoring data are initially screened to identify any detection above the modeled estimated concentrations and to determine if exposure pathways not previously identified as routes of exposure in the environment are possible. The key elements (questions) to consider for all sample media are:

* Relevance
	+ How did sampling locations compare to the pesticide use location(s) including proximity and/or vulnerability (*e.g*., leaching and runoff)?
	+ Were samples collected at a time when offsite transport (*e.g*., runoff, leaching, spray drift, and volatilization) was likely?
	+ What were the weather patterns (*e.g*., rainfall) before, during, and after pesticide application?
	+ Was the sample media relevant to the habitats of concern?
	+ Was the study a field-scale[[13]](#footnote-13) or general monitoring study[[14]](#footnote-14)?
	+ Did the pesticide use pattern reflect the pesticide use being assessed?
* Methods
	+ How often were samples collected? was it frequent enough to estimate the desired duration of exposure?
	+ Were the detection limits and limits of quantitation reported and were method recoveries reported sufficient to have confidence in the results?
	+ Were the detection limits and limits of quantitation reported consistent with the needs of the assessment (*e.g.*, is the detection limit below the toxicological endpoint of concern)?
	+ How often were samples collected? Was the sample frequency enough to estimate the desired duration of exposure?
	+ What was the type of sample? Was the sample a single point (often called grab), integrated, or composite (with respect to time, depth, distance, or individuals) sample?
	+ What quality assurance and quality control measures were utilized?

Additional sample media specific elements to be considered include:

* Surface Water
	+ For agricultural areas, were the samples collected from sites where the pesticide was used?
	+ For urban areas, were the samples collected from sites receiving surface water runoff via stormwater conveyances?
	+ How did runoff vulnerability at the sampling sites compare to the overall pesticide use area?
	+ If composite sample, was it a time, depth, or flow-weighted sample (need stream hydrograph)?
* Groundwater
	+ Was there a pathway between the use site and the aquifer sampled?
	+ Has sufficient time elapsed between the pesticide application and sampling event for the chemicals to have leached through the soil profile to the sampled well?
	+ Was the groundwater sample taken from confined or unconfined aquifers?
	+ What was the well depth and at what depth was the well screened?
	+ What was the type of well (*e.g*., irrigation, drinking water, observation, etc.)?
* Sediment
	+ What were the sediment characteristics (*e.g.*, organic carbon/matter content, grain size, and redox potential)?
	+ What was the disturbance regime?
* Air
	+ Was the monitoring study design (*e.g*., sampler locations) appropriate considering land configuration, terrain, and meteorological variations?
* Precipitation
	+ What was the distance between the pesticide use area and the sample site?
* Biota
	+ How was the specimen collected?
	+ Why was the specimen collected?
	+ Were metabolites analyzed?
	+ How was the specimen characterized [*e.g.*, species, age class, condition, external deformities, erosion, lesions, and Tumors (DELTs), weight, length].

## Use of Monitoring Data for Risk Assessment Purposes

For ESA evaluation of pesticides, general monitoring studies that provide information on pesticide concentrations based on monitoring of specific locations at specific times but are not associated with field-scale monitoring of specific applications of pesticides under well-described conditions should not be used for quantitative comparison to thresholds (NAS, 2013[[15]](#footnote-15)). Commonly, monitoring data are used qualitatively in risk assessments for characterization (*e.g*., line of evidence in a weight-of-evidence approach or determine if downstream transport of pesticides could occur to the species range or critical habitat). For most pesticides, monitoring data are typically insufficient for quantitative use under all potential use conditions and geographic scales. The extent to which monitoring data can be used to establish a reasonable high end exposure estimate for a specific exposure scenario depends on how much is known about the ancillary data, the robustness of the dataset, and the extent to which the data represent the exposure scenario of interest, and the likelihood that the sampling regime included sampling during the occurrence of the peak environmental concentration. For example, the risk assessor should consider the adequacy of the data based on how well the study is coordinated with pesticide applications, the frequency and number of years of sampling, the quality and contents of the ancillary data, and the ability to correlate this information to the detections and the pesticide use pattern being evaluated.

Each data source should be adequately characterized, including temporal and spatial characterization as well as method detection limits and a summary of the results (*e.g*., number of samples analyzed, number of sites, site characterization, number of detections, detected concentrations, sample frequency, trends over time).

### Quantitative Use of Monitoring Data for Risk Assessment Purposes

The quantitative use of monitoring data (*i.e.*, reasonable upper bound exposure concentration) for risk assessment purposes occurs infrequently. Available monitoring data are typically not coordinated with a pesticide application or are of insufficient frequency to capture durations of exposure concern (*e.g*., peak, 24-hour, 4-day) and are also temporally and spatially limited. Yet, monitoring data with adequate ancillary data and study design/objectives may be used quantitatively. Most commonly, monitoring data are used quantitatively when there are no methods available for modeling exposure or on a site-specific basis when monitoring data are available for a particular location. Quantitative use of monitoring data for risk assessment purposes includes the use of a measured concentration or an adjusted value [a measured concentration value adjusted (*i.e*., application of a bias factor[[16]](#footnote-16),[[17]](#footnote-17)) to reflect uncertainties or inadequacies of the monitoring data to generate an exposure value] as a direct measure of exposure. Monitoring data are more likely to be used quantitatively as a local refinement given uncertainties in extrapolating results to other locations.

###  Use of Monitoring Data for Evaluation of Downstream Transport

After a species has been classified as an No Effect or Not Likely to Adversely Affect, EPA evaluated monitoring data upstream and downstream of the species range/critical habitat to determine if any detections of the pesticide had occurred and ensure that no sources upstream of a species range or critical habitat would affect the species.

To do this analysis, EPA first used Esri ArcGIS tools to identify streams (NHDPlus Version 2[[18]](#footnote-18)) that crossed the boundary of the species range/critical habitat. EPA then used the latitude and longitude data for monitoring sites obtained from the Water Quality Portal (<https://www.waterqualitydata.us/portal/>) to index the sites to NHDPlus stream segments. EPA developed Python scripts that utilize NHDPlus to identify monitoring sites that hydrologically connected to each species range/critical habitat and provide a corresponding upstream/downstream distance and hydrologic travel time between the monitoring site and the range/critical habitat. EPA then categorized the connected monitoring sites into three areas: those sites that occurred within the borders of the species range/critical habitat; those sites that occurred within 68-stream miles upstream of the species range/critical habitat; and those sites that occurred greater than 68-stream miles upstream of the species range/critical habitat. The 68-stream mile limit was initially used to identify those sites that were within a 1-day travel time of the species range/critical habitat. Only upstream locations were categorized for the analysis as there is uncertainty in the downstream monitoring sites as to where the pesticide originated. Given the chemical’s persistence, it would also be important to evaluate sites beyond this distance, as the pesticide still might reach the species range/critical habitat.

###  Qualitative Use of Monitoring Data for Risk Assessment Purposes

If available monitoring data do not meet the standards for quantitative risk assessment use, it does not mean the data are not useful for risk assessment. Available monitoring data may still be valuable in adding context to the exposure assessments depending on the available ancillary data. For instance, detections of a given pesticide can provide a measure of a lower bound of exposure based on actual use or trends over time. While the data may not be sufficiently robust to ensure a high-end exposure has been observed, detected concentrations can confirm that pesticide(s) may be transported off a treated field. In addition, identification of pesticide co-occurrences (environmental mixtures) as well as the proximity of detections to species locations are useful information to include in characterizing the potential effects of pesticide use on non-target taxa. A comparison of model-estimated and measured pesticide concentrations may be useful for characterization; however, unless the model is parameterized to simulate specific sampling sites, the degree of over- or under-prediction should be limited as model simulations are intended to be inclusive of vulnerable scenarios and are expected to be greater than measured concentrations. Nonetheless, this information may be used as part of a weight-of-evidence approach to support potential exposure pathways as well as exposure concentrations.

Monitoring data that provide no context on where samples are taken, what the study objectives are, what analytical methods are used, or what the detection limits are, should be mentioned; however, these data should not be used in a qualitative manner. When examining multiple sources of monitoring data, it is also important consider the potential for duplication (*i.e.*, monitoring data reported for the sample study/sample in multiple databases).

### Future Enhancements in Quantitative Use of Monitoring Data for Risk Assessment Purposes

EPA is investigating several tools that would allow for the consideration of using less robust monitoring datasets to inform the various lines of evidence being used to evaluate aquatic exposure. These tools have been recently evaluated with regards to their use in drinking water assessments[[19]](#footnote-19), but have not yet been vetted for use in ecological risk assessments or presented to the Services, so this investigation is still considered preliminary.

Sampling bias factors could be used to adjust monitoring data and allow for an estimation of high-end exposure. The approach involves bootstrapping simulations of sampling frequencies to develop simple multiplicative factors for exposure estimates. The uncertainty of different sampling frequencies in estimating exposures of varying durations is characterized. For short-term sampling bias factors (i.e., durations less than 28 days), using various defined sampling windows (4 to 28-days) across a robust monitoring dataset, a random day within each sampling window is selected to simulate a monitoring event, and then a daily chemograph is generated, with the process being repeated 10,000 times. For each of these time series, the 1-day peak and maximum rolling average for each of the averaging periods is calculated. For long-term sampling bias factors (*i.e*., annual concentrations and longer), the upper confidence limit around the mean for a number of randomly selected samples is estimated, with the process being repeated 10,000 times. A sampling bias factor is then calculated by comparing the 5th percentile of the estimated maximums from the simulations to the actual maximums. The sampling bias factor then becomes a multiplicative factor that can be applied to an exposure estimate, depending on the sampling frequency and the duration of exposure.

Another tool being explored is the SEAWAVE-QEX model, developed by the U.S. Geological Survey (Vecchia, 2018[[20]](#footnote-20)). In 2013, the U.S. Geological Survey released SEAWAVEQ, a regression model which generates an equation relating pesticide concentration to the seasonal wave of variability between concentrations and flow. Using consecutive years of pesticide monitoring data along with daily stream flow that corresponds to the pesticide sampling period, SEAWAVEQ predicts the trend in pesticide concentrations over time, capturing concentrations that may be higher than those observed in monitoring.

In 2018, the U.S. Geological Survey released SEAWAVE-QEX to address recommendations from previous SAPs. SEAWAVE-QEX is a time series model that can develop daily pesticide chemographs from non-daily sampling data, using streamflow or other covariates specific to the sampling site. Like its predecessor SEAWAVE-Q, SEAWAVE-QEX incorporates measured pesticide concentrations, seasonality, streamflow variability, and long-term trends into its simulations. However, SEAWAVE-QEX is further refined to approximate non-constant (seasonal) variance of model residuals and serial correlation between measured concentrations. The model can also generate multiple, equally probable simulated chemographs. SEAWAVE-QEX captures the seasonal nature of pesticide applications as well as the sporadic nature of application events and runoff events, and specifically addresses the concerns expressed by previous SAPs regarding the consistent underestimation of pesticide concentrations occurring between sampling events from other infilling methods.

1. Young, D.F. and Fry, M.M., 2014. A Model for Predicting Pesticide in Runoff, Erosion, and Leachate: User Manual, U.S. Environmental Protection Agency, Washington, DC. USEPA/OPP 734F14002. [↑](#footnote-ref-1)
2. Young, D. F., 2014. The Variable Volume Water Model, U.S. Environmental Protection Agency, Washington, DC. USEPA/OPP 734F14003. [↑](#footnote-ref-2)
3. Available at: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-3)
4. Available: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment#aquatic> [↑](#footnote-ref-4)
5. USDA, 1986. Urban Hydrology for Small Watersheds, TR-55. United States Department of Agriculture, Technical Release 55. Natural Resources Conservation Service. Available: http://www.cpesc.org/reference/tr55.pdf [↑](#footnote-ref-5)
6. Han, W., Yang, Z., Di, L., Yue, P., 2014. A geospatial Web service approach for creating on-demand Cropland Data Layer thematic maps. Transactions of the ASABE, 57(1), 239-247.

Available: http://www.nass.usda.gov/Research\_and\_Science/Cropland/SARS1a.php [↑](#footnote-ref-6)
7. Young, D., 2013. Pesticides in Flooded Applications Model (PFAM): Conceptualization, Development, Evaluation, and User Guide, EPA-734-R-13-001. Available: http://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100LE7H.txt [↑](#footnote-ref-7)
8. USDA, NASS Census of Agriculture. 2012. <https://www.nass.usda.gov/Publications/AgCensus/2012/#full_report> [↑](#footnote-ref-8)
9. NOAA National Climatic Data Center, 1993. Solar and Meteorological Surface Observation Network (SAMSON) 1961-1990, Version 1.0, Sep 1993. Available: http://www2.epa.gov/exposure-assessment-models/meteorological-data [↑](#footnote-ref-9)
10. NAS, 2013. Assessing Risks to Endangered and Threatened Species from Pesticides. The National Academies Press. 2013 [↑](#footnote-ref-10)
11. Available: <http://www.ipmcenters.org/cropprofiles/> [↑](#footnote-ref-11)
12. Available: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-12)
13. A field-scale monitoring study is defined as the monitoring of specific applications of the pesticide at the field-scale under well-described conditions. These studies provide the information needed to make direct comparison to exposure model estimates including the rate, method, treatment location relative to monitoring stations, meteorological conditions, characteristics of the treated site, and characteristics of the habitat sampled. [↑](#footnote-ref-13)
14. A general monitoring study is defined as a study that provides information on pesticide concentrations in the environment at specific locations and times. Sampling may also be coordinated with the use of pesticides at some level (e.g., at the watershed level), but are typically not coordinated with specific applications of pesticides or the meteorological conditions of the habitat. [↑](#footnote-ref-14)
15. NAS, op. cit., p.14 [↑](#footnote-ref-15)
16. For example, even in the case of a pesticide such as atrazine, the sample frequency should be daily, otherwise the Federal Insecticide, Fungicide, and Rodenticide Act Science Advisory Panel referenced in footnote 10 below recommends a sampling bias factor be applied to monitoring data to develop upper bound exposure concentrations. [↑](#footnote-ref-16)
17. U.S. Environmental Protection Agency. Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) Scientific Advisory Panel Meeting: Problem Formulation for the Reassessment of Ecological Risks from the Use of Atrazine, **June 12-14, 2012**, Docket Number: EPA-HQ-OPP-2012-0230 [↑](#footnote-ref-17)
18. <http://www.horizon-systems.com/nhdplus/> [↑](#footnote-ref-18)
19. Approaches for Quantitative Use of Surface Water Monitoring Data in Pesticide Drinking Water Assessments, November 19-22, 2019, <https://www.epa.gov/sap/meeting-information-november-19-22-2019-scientific-advisory-panel> [↑](#footnote-ref-19)
20. Vecchia, A.V., 2018. Model methodology for estimating pesticide concentration extremes based on sparse monitoring data: U.S. Geological Survey Scientific Investigations Report 2017–5159, 47 p. Available at https://doi.org/10.3133/sir20175159. and other ancillary variables: U.S. Geological Survey Open-File Report 2013–1255, <http://dx.doi.org/10.3133/ofr20131255> [↑](#footnote-ref-20)