## Attachment 3-1: Background Document: Aquatic Exposure Estimation for Endangered Species

# Executive Summary

The U.S. Environmental Protection Agency, Office of Pesticide Programs, in partnership with the Fish and Wildlife Service, the National Marine Fisheries Service, and the U.S. Department of Agriculture, has 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 are generated based on key fate and transport processes, using chemical and application information, soil parameters, and watershed and water body characteristics. Recommendations for improving upon these methods from stakeholders, the scientific community, and the public are welcome and encouraged.

The following information is organized to discuss the revised conceptual model, modeling components, issues modeling medium and high flowing surface water bodies, and the use of monitoring data for endangered species assessment (ESA) Biological Evaluations (BEs) for chlorpyrifos, diazinon, and malathion.

**Contents**

[1. Executive Summary 1](#_Toc471823795)

[2. Introduction 4](#_Toc471823796)

[3. Aquatic Modeling 4](#_Toc471823797)

[3.1. Surface Water Modeling 4](#_Toc471823801)

[3.1.1. Traditional Approach 4](#_Toc471823806)

[3.1.2. Revised Conceptual Model and Approach for ESA 6](#_Toc471823807)

[3.1.3. Modeling Components 8](#_Toc471823814)

[3.1.3.1. Input Scenarios 8](#_Toc471823815)

[3.1.3.1.1. Regional Spatial Delineation for Scenarios 10](#_Toc471823824)

[3.1.3.1.2. Association to Agricultural and Nonagricultural Data Layers 10](#_Toc471823825)

[3.1.3.1.3. Development of Representative Scenarios 13](#_Toc471823826)

[3.1.3.1.4. Spatial Delineation, Weather Data 13](#_Toc471823827)

[3.1.3.2. Aquatic Habitat Bins 16](#_Toc471823828)

[3.1.3.3. Watershed Sizes 19](#_Toc471823829)

[3.1.3.3.1. Flowing Bins 19](#_Toc471823830)

[3.1.3.3.2. Static Bins 21](#_Toc471823831)

[3.1.3.3.3. Estuarine and Marine 22](#_Toc471823844)

[3.1.3.4. Application Date Selection 25](#_Toc471823845)

[3.1.3.5. Spray Drift Exposure 26](#_Toc471823857)

[3.1.4. Issues Modeling Medium- and High-Flowing Waterbodies 27](#_Toc471823865)

[3.1.4.1. Overview of Issues 27](#_Toc471823866)

[3.1.4.2. Modifications to Modeling Approach 28](#_Toc471823876)

[3.1.4.2.1. Modifications Considered But Not Incorporated 29](#_Toc471823877)

[3.1.4.2.1.1. Incorporation of Base Flow 29](#_Toc471823878)

[3.1.4.2.1.2. Percent Use Area and Percent Use Treatment Adjustment Factors 32](#_Toc471823891)

[3.1.4.2.1.3. Adjustment of Water Body Length 32](#_Toc471823905)

[3.1.4.2.1.4. Spreading Out Applications 33](#_Toc471823906)

[3.1.4.2.2. Modifications Explored and Incorporated into Modeling 33](#_Toc471823907)

[3.1.4.2.2.1. Curve Number Adjustment 33](#_Toc471823908)

[3.1.4.2.2.2. Daily Flow Averaging 35](#_Toc471823909)

[3.1.4.2.2.3. Adjustment of Water Body Dimensions 35](#_Toc471823910)

[3.1.4.2.2.4. Use of Daily Average EEC 36](#_Toc471823911)

[3.1.4.3. Modifications Evaluation, Case Study and Results 36](#_Toc471823912)

[3.1.4.4. Modifications Evaluation, Pilot Chemicals, Draft BEs 43](#_Toc471823913)

[3.1.4.5. Modifications Evaluation, Pilot Chemicals, Final BEs 44](#_Toc471823926)

[3.2. Pesticide Flooded Application Model (PFAM) 46](#_Toc471823927)

[4. Use of Monitoring Data 47](#_Toc471823928)

[4.1. Evaluation of Monitoring Data 48](#_Toc471823930)

[4.2. Use of Monitoring Data for Risk Assessment Purposes 50](#_Toc471823931)

[4.2.1. Quantitative Use of Monitoring Data for Risk Assessment Purposes 50](#_Toc471823932)

[4.2.2. Qualitative Use of Monitoring Data for Risk Assessment Purposes 51](#_Toc471823933)

[4.2.3. Future Enhancements in Quantitative Use of Monitoring Data for Risk Assessment Purposes 51](#_Toc471823934)

# Introduction

Methods and modeling techniques have been developed to estimate pesticide aquatic exposure concentrations for endangered species for use in the biological evaluation (BEs) for chlorpyrifos, diazinon, and malathion. 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.*, humans, 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, several refinements to the assumed conditions and aquatic exposure pathways (water bodies) (**Table A 3-1.1**) 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



## 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, version 1.52, May 2016). 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 (a “Standard Pond” for aquatic organisms or an “Index Reservoir” for drinking water) receiving the pesticide loading by runoff, erosion, and spray drift from the field. For the endangered species assessments, PRZM5 is applied in the same way and the VVWM has been extended to simulate a range of surface water bodies and 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.

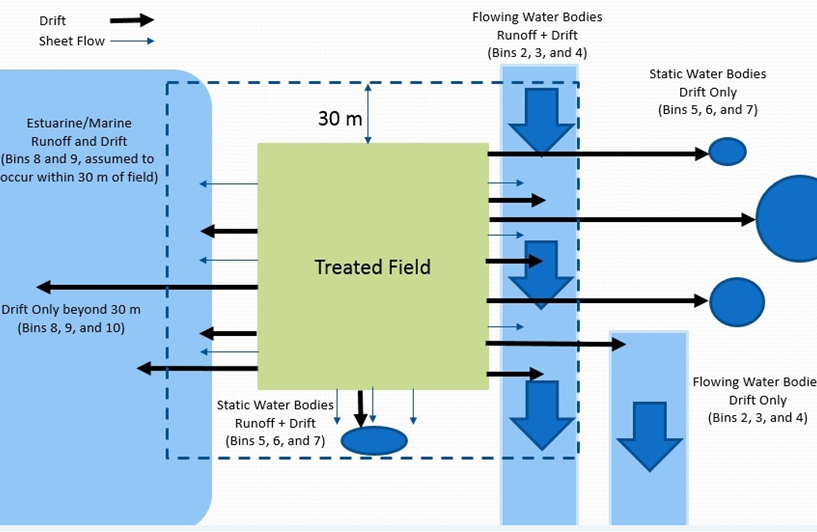
**Table A 3-1.1**. **Aquatic modeling for ecological assessments and the endangered species biological evaluation refinements**

| **Model Component/Process** | **Current Aquatic Modeling for Ecological Assessments** | **Endangered Species Assessment Refinements** |
| --- | --- | --- |
| Catchment area | 10 ha (25 ac) agricultural field | Defined based on methods in **Section 3.1.3.3** |
| Catchment soil conditions | Single, runoff prone (Hydrologic Soil Group C or D) soil type for entire field or watershed.  Runoff driven by curve numbers (crop and no crop) that represents the single soil and crop use being modeled. | Same |
| Pesticide inputs into catchment | Application according to label rates and timing, adjusted for crop area (assumes 100% of field for exposure) | Same |
| Pesticide fate in catchment (and amount available for transport) | First-order transformation and linear equilibrium sorption in soil.  Finite difference solution to advection-dispersion equation. | Same |
| Weather inputs | 30 years (1961-1990) (SAMSON dataset) | Same |
| Water body | Standard Pond- 1 ha (2.5 acres) x 2 m deep | 10 habitat bins (see **Table B 3-1.2**) |
| Pesticide inputs to water | Pesticide mass flux in runoff (dissolved) and erosion (sorbed) by rain events.  Spray drift mass based on application. | Same |
| Pesticide fate in water | Aerobic aquatic half-life (metabolism, hydrolysis, photolysis).  First-order mass transfer between water column and sediment.  Equilibrium partitioning to sediment | Same |
| Water body flow/dilution | Pesticide mass added instantaneously to fixed water body volume.  No flow in Standard Pond (static) | Downstream dilution may be used from the edge of the use area, which consists of a percent use area adjustment. Concentrations are reduced by the use area adjustment factor until concentrations are below levels of concern. |

### Revised Conceptual Model and Approach for ESA

Building upon the existing ecological exposure modeling framework (**Section 3.1.1**), this modified approach for ESA delineates additional water body types (or habitats) to characterize a range of potential exposures to endangered species. **Figure B 3-1.1** (**Table A 3-1.2**) summarizes the various aquatic habitat bins that have been developed, in place of the single, Standard Pond, to evaluate exposure in static and flowing freshwater bodies and estuarine/marine water bodies. Within 30 meters, all flowing and static aquatic bins, as well as intertidal and subtidal nearshore bins, would receive runoff and spray drift. The 30-meter distance was considered the maximum distance where runoff would occur as sheet flow[[4]](#footnote-4). Beyond this distance, concentrated flow would be assumed and concentrations would be assessed using the flowing bins and the downstream dilution tool. Beyond 30 meters, aquatic bins would only receive spray drift.

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 A 3-1.1**. **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 A 3-1.2. Endangered species aquatic habitat bins**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Generic Habitat | Depth (meters) | Width (meters) | Length (meters) | Flow (m3/second) |
| 1 – Aquatic-associated terrestrial habitats1 | NA | NA | NA | NA |
| 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 Bin 1 does not have dimensions like the other 9 bins; different methods are used to evaluate exposure.

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



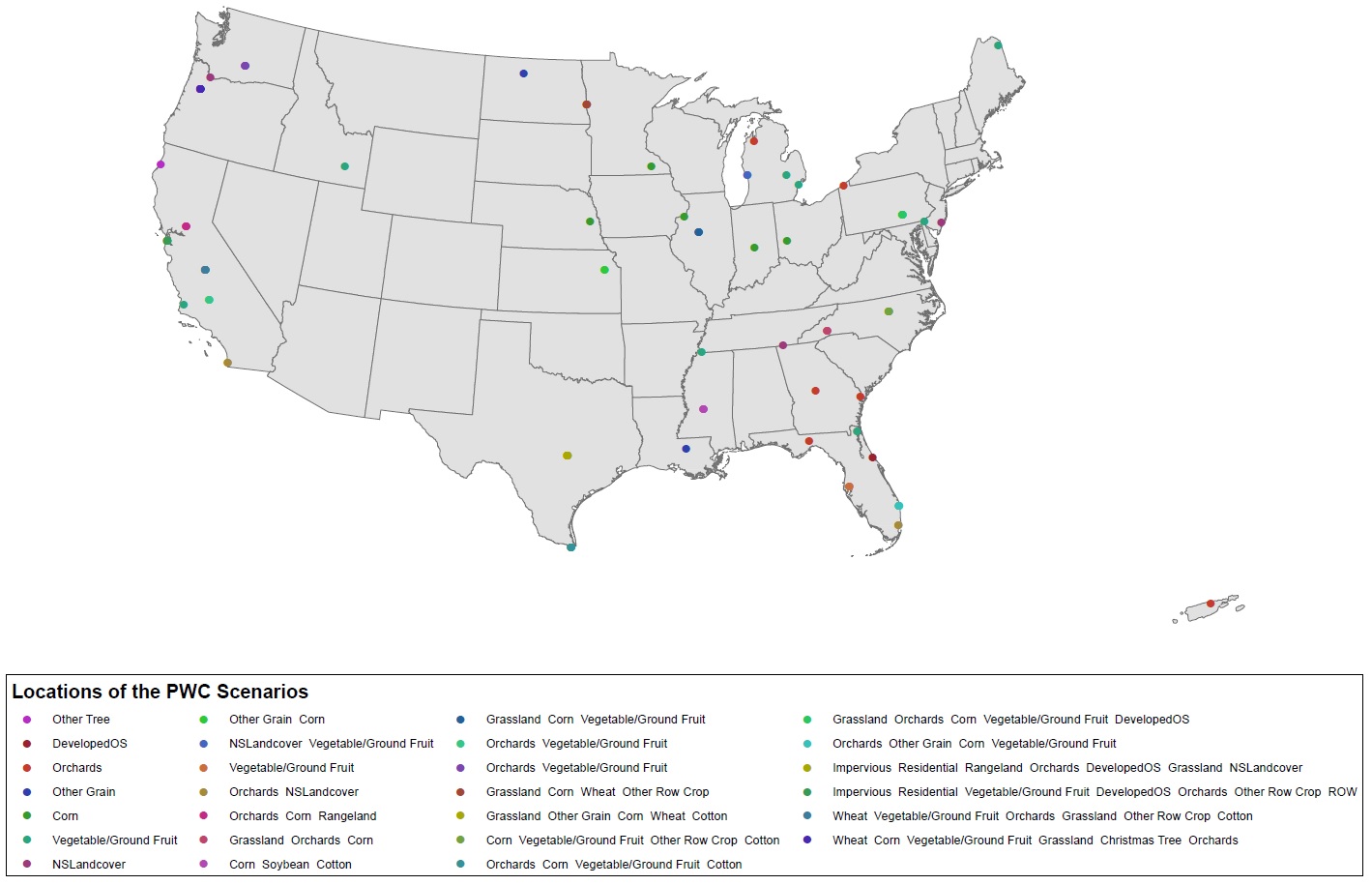
### Modeling Components

#### 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. There is a large suite of existing surface water scenarios available (123 total) for use in PRZM5/VVWM simulations, spanning a range of agricultural and non-agricultural pesticide use sites.[[5]](#footnote-5) The locations of the existing scenarios are shown in **Figure A 3-1.2**. However, there are instances when a scenario does not exist for a particular crop 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 may be modeled with a surrogate scenario using one of two approaches. In the first approach, the scenario may be 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[[6]](#footnote-6)), 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).

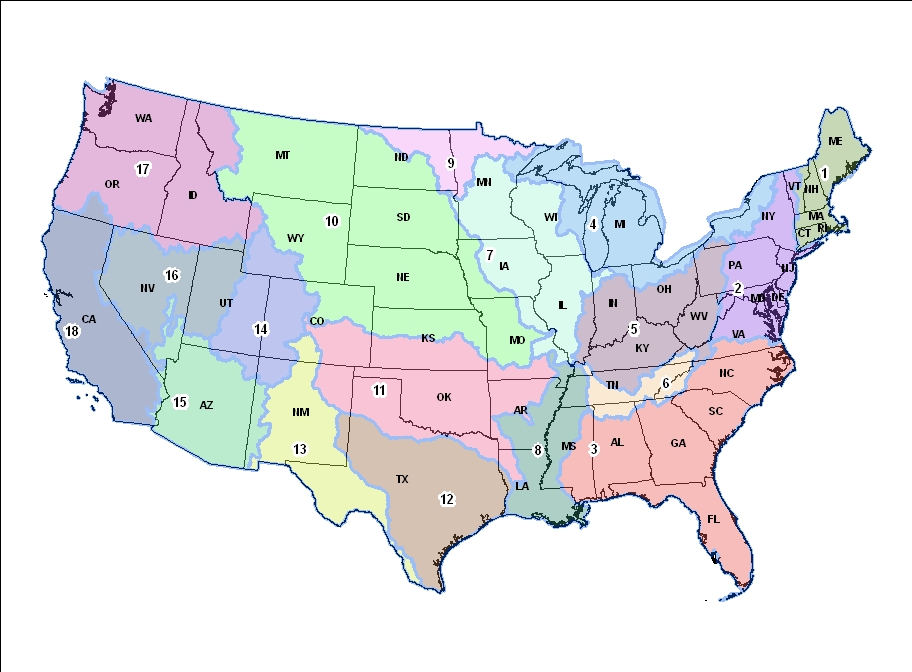
 **Figure B 3-1.2. Location of existing aquatic exposure modeling scenarios.**

A matrix was developed to assign one input scenario per hydrologic unit code 2 (HUC2) region and crop group combination (**Table A 3-1.3**). The following steps were completed to select the representative scenario (including the weather station) for each HUC2 region-crop group combination.



##### Regional Spatial Delineation for Scenarios

HUC2 regions are used as the geospatial reference for scenario selection (**Figure A 3-1.3**).



**Figure A 3-1.3. Spatial distribution of HUC2 regions and U.S. state boundaries**

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

The crop group for each scenario is based on the USDA National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL)[[7]](#footnote-7), which offers annual, geospatially referenced crop-specific land cover information from satellite imagery. Using Geographical Information System (GIS) software, the HUC2 regions are overlaid with the USDA CDL to identify the cropped areas (in acres) within each HUC2 region. Five CDL years (2010-2014) have been temporally aggregated, and the 111 crop categories native to CDL have been 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. Rice is 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)[[8]](#footnote-8), separate from the PRZM5/VVWM ESA scenarios described here (**Section 3.1.3.1**).

The NASS Agricultural Census data has been used to confirm growing regions for each crop group. If any crops are identified in the Agricultural Census that are not otherwise identified within a HUC2 region based on the CDL data, an input scenario is assigned for the corresponding HUC2 region-crop group combination. The results of this analysis are presented in **Table A 3-1.3**. Cotton, orchards and vineyards, and other trees are the only crop groups identified with no acreage within certain HUC2s. Based on this analysis, the HUC2 region-crop group combinations that have no acreage are excluded from the scenario selection process (identified in **Table A 3-1.3** as blacked out cells). If a small acreage is noted for a HUC2 region-crop group combination, a representative or surrogate scenario is identified.

Ten nonagricultural uses also have been 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 are identified for a particular HUC2 region, an “a” or “b” is added to the scenario name.

**Table A 3-1.3. Crop acres, by Crop Data Layer category and HUC2 region**

|  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC 2**  **Region** | **Corn** | **Soybean** | **Cotton** | **Pasture/**  **Hay** | **Other Crops** | **Orchards and Vineyards** | **Other Trees** | **Other Grains** | **Other Row Crops** | **Wheat** | **Vegetables and Ground Fruit** | **Rice** |
| **01** | 358,677 | 13,619 | 0 | 2,299,065 | 114,592 | 43,296 | 4,170 | 184,053 | 8,590 | 7,021 | 396,400 | 0 |
| **02** | 6,437,058 | 5,036,696 | 59,213 | 12,092,087 | 1,532,024 | 379,082 | 6,1976 | 983,442 | 13,301 | 2,700,573 | 363,841 | 1 |
| **03** | 7,239,956 | 8,899,772 | 9,269,747 | 39,422,125 | 7,788,600 | 2,469,350 | 1,771 | 1,731,680 | 4,197,455 | 5,323,073 | 570,695 | 24,764 |
| **04** | 17,907,319 | 14,059,702 | 0 | 16,329,022 | 1,227,385 | 959,467 | 62,905 | 1,046,561 | 704,904 | 5,904,198 | 1,864,904 | 1 |
| **05** | 22,682,226 | 22,162,761 | 4,721 | 24,208,197 | 642,247 | 65,581 | 16,635 | 248,699 | 46,590 | 5,034,839 | 288,184 | 33 |
| **06** | 1,193,082 | 1,242,864 | 524,068 | 7,185,211 | 63,565 | 30,406 | 9,043 | 21,400 | 9,358 | 620,656 | 32,212 | 1 |
| **07** | 57,748,484 | 55,163,033 | 687 | 29,514,129 | 253,889 | 71,815 | 2,810 | 1,497,659 | 493,776 | 5,310,065 | 1,788,603 | 5,395 |
| **08** | 8,813,986 | 17,114,672 | 9,837,633 | 8,081,052 | 4,653,753 | 147,194 | 0 | 1,978,115 | 61,285 | 6,191,869 | 133,488 | 7,344,580 |
| **09** | 7,663,065 | 12,777,001 | 0 | 9,971,751 | 2,000,936 | 0 | 0 | 4,237,298 | 4,150,689 | 16,916,992 | 2,857,719 | 0 |
| **10** | 58,577,416 | 46,412,519 | 348 | 197,647,865 | 22,118,235 | 45,888 | 373 | 15,313,057 | 5,647,024 | 53,147,405 | 4,873,417 | 11 |
| **11** | 11,325,835 | 7,112,783 | 3,419,503 | 84,690,678 | 10,819,174 | 225,280 | 7 | 11,297,418 | 329,644 | 29,754,728 | 127,076 | 889,374 |
| **12** | 4,227,885 | 449,521 | 11,049,544 | 45,393,414 | 5,585,025 | 509,541 | 460 | 7,034,185 | 319,888 | 7,812,459 | 96,428 | 795,211 |
| **13** | 159,088 | 3,287 | 329,819 | 20,898,576 | 1,070,392 | 247,804 | 336 | 508,002 | 19,513 | 261,299 | 241,345 | 4 |
| **14** | 206,984 | 373 | 61 | 11,069,850 | 399,163 | 33,791 | 0 | 198,050 | 18,472 | 358,150 | 175,998 | 0 |
| **15** | 211,196 | 8 | 1,001,954 | 7,409,916 | 2,087,833 | 112,291 | 0 | 381,894 | 500 | 433,978 | 244,647 | 0 |
| **16** | 314,238 | 45 | 0 | 9,306,753 | 1,011,709 | 40,280 | 127 | 588,750 | 746 | 919,038 | 80,873 | 0 |
| **17** | 1,990,248 | 5,601 | 0 | 36,862,718 | 6,271,300 | 1,607,201 | 109,830 | 2,680,486 | 823,676 | 11,674,036 | 4,203,380 | 0 |
| **18** | 2,634,163 | 68 | 2,656,390 | 26,638,360 | 4,184,381 | 6,719,251 | 0 | 1,958,542 | 357,380 | 2,708,716 | 1,829,285 | 1,161,516 |
| **19** |  |  |  | 42,437 |  | 15 |  | 5,348 |  | 182 | 299 |  |
| **20** | 8,374 | 0\* | 0 | 3,456 | 504 | 120,697 | 0\* | 7 | 54 | 0\* | 13,983 | 0 |
| **21** | 1,026 | 0 | 0 | 0 | 0 | 49,007 | 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 are binned based on location and crop into HUC2 region-crop group combinations. The scenario with the highest runoff curve number is identified per HUC2 region-crop group combination, as it represents the highest runoff potential. For those HUC2 region-crop group combinations where input scenarios are not available, a surrogate scenario (with the highest runoff potential) from a neighboring HUC2 region is selected.

**Table A 3-1.4** 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 is used. For impervious and residential uses, the CAImperviousRLF and CAresidentialRLF scenarios are 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 scenarios developed, a representative SAMSON weather station is assigned from among the stations located within the corresponding HUC2 region, based on the highest 30-year rainfall level (**Table A 3-1.5**).

In order to identify the representative station for use with the scenarios, the 242 meteorological stations are grouped by HUC 2 and cumulative 30-year precipitation value is 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 is a large difference in the precipitation values (*i.e.*, the maximum cumulative 30-year precipitation value for a HUC2 is three times greater than the minimum value). For HUC2 regions where a large rainfall difference occurs, the median precipitation value is used as a demarcation between a high-precipitation and low-precipitation group. The median station for both the high-precipitation and the low-precipitation groups are identified as representative weather stations and two sets of modeling is 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 exists between the highest precipitation station and remaining stations in the HUC2 (*i.e.*, precipitation value at least twice as high as the median of the other stations). For these HUC2 regions, the highest precipitation weather station is selected along with the weather station with the median cumulative 30-year precipitation value for the remaining stations.

**Table A 3-1.4. PRZM5/VVWM surrogate scenarios used for ESA aquatic modeling**

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

**Table A 3-1.4. PRZM5/VVWM surrogate scenarios used for ESA aquatic modeling (continued)**

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC 2** | **Non-specified land cover** | **Orchards/**  **Vineyards** | **Other Trees**  **Christmas Tree1** | **Other Grain** | **Other Row Crop** | **Wheat** | **Vegetables/**  **Ground Fruit** |
| **01** | MInurserySTD | NYgrapesSTD | NYgrapesSTD | PAalfalfaOP | MEpotatoSTD | PAalfalfaOP | MEpotatoSTD |
| **02** | NJnurserySTD\_V2 | PAapplesSTD\_V2 | PAapplesSTD\_V2 | PAalfalfaOP | NJmelonSTD | PAalfalfaOP | PAvegetableNMC |
| **03** | FLnurserySTD\_V2 | FLcitusSTD | FLcitusSTD | FLsugarcaneSTD | NCpeanutSTD | NCalfalfaOP | FLpotatoNMC |
| **04** | MInurserySTD | NYgrapesSTD | 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 | FLcitusSTD | FLcitusSTD | ILalfalfaNMC | ILcornSTD | ILalfalfaNMC | ILbeansNMC |
| **08** | FLnurserySTD\_V2 | FLcitusSTD | FLcitusSTD | LAsurgarcaneSTD | MOmelonSTD | ILalfalfaNMC | MOmelonSTD |
| **09** | MInurserySTD | NYgrapesSTD | MIcherriesSTD | NDcanolaSTD | Mnsugarbeet | NDwheatSTD | MNsugarbeatSTD |
| **10** | TNnurserySTD\_v2 | ORFilbert | FLcitusSTD | KSsorghumSTD | KScorn | NDwheatSTD | MNsugarbeatSTD |
| **11** | TNnurserySTD\_v2 | OrchardBSS | FLcitusSTD | TXwheatOP | NECornSTD | TXwheatOP | STXmelonNMC |
| **12** | NurseryBSS\_V2 | OrchardBSS | OrchardBSS | TXwheatOP | STXcornNMC | TXwheatOP | STXmelonNMC |
| **13** | NurseryBSS\_V2 | OrchardBSS | OrchardBSS | TXwheatOP | STXcornNMC | TXwheatOP | STXmelonNMC |
| **14** | TNnurserySTD\_v2 | OrchardBSS | OrchardBSS | TXwheatOP | STXcornNMC | TXwheatOP | STXmelonNMC |
| **15** | NurseryBSS\_V2 | CAcitrus\_WirrigSTD | CAcitrus\_WirrigSTD | TXwheatOP | STXcornNMC | TXwheatOP | CALettuceSTD |
| **16** | CAnurserySTDV | CAcitrus\_WirrigSTD | CAcitrus\_WirrigSTD | TXwheatOP | STXcornNMC | TXwheatOP | 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 | FLcitrusSTD | FLcitrusSTD | FLsugarcaneSTD | FLpotatoNMC |  | FLtomatoSTD |
| **21** | FLnurserySTD\_V2 | PRCoffee | PRCoffee |  |  |  | FLtomatoSTD |

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

**Table A 3-1.5. Representative weather stations by HUC2 region1**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| 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 A 3-1.2**). The nine aquatic habitat bins (or water body types) are used in aquatic exposure modeling and 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.

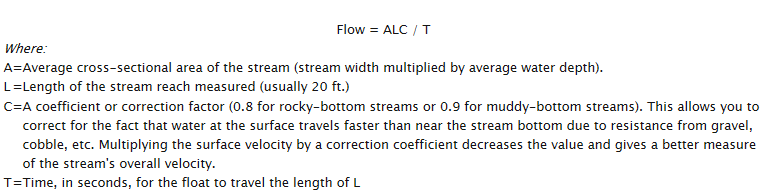
Each habitat bin has been developed to represent three general categories: freshwater static waters, freshwater flowing waters, and estuarine/marine waters. For each bin, representative dimensions and flow regimes are defined (**Table A 3-1.2**). The rationale used to develop each habitat bin is described below.

**Terrestrial.** Terrestrial habitats include both upland and aquatic-associated habitats. Terrestrial habitats will require separate binning consideration or other analysis and are not addressed further in this document. Terrestrial habitats and exposure are discussed in **Section 3.3** of each of the pilot chemical BEs. 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 are 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.

**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 seem reasonable to estimate concentration ranges in flowing water habitats. Distinctions were made based on the range in flow rates. Flow rates take into account 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 below 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, the Standard Pond 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.* 20,000,000 liters). For the ESA aquatic habitat bins, watershed sizes are similarly defined for each bin. For flowing water bodies, the watershed area is estimated using a regression of drainage area to annual average flow, based on the National Hydrography Dataset version 2 (NHDPlus v2) (McKay et al., 2012[[11]](#footnote-11)). For static waters, an appropriate watershed size is selected to ensure that runoff volumes do not excessively overflow the static water volume. Watershed area estimates are described below for the various bins.

##### Flowing Bins

To determine the watershed area for flowing water body bins, a regression analysis is used to define the relationship between drainage area and annual average flow rates (**Table A 3-1.6**). **Figure A 3-1.4** provides an example of the regression for HUC2 region 1. Using the NHDPlus v2 dataset, the following steps were taken:

* For each NHDPlus region (which corresponds to HUC2 regions), the Enhanced Runoff Method (EROM) data are joined to the NHD flowline data.
* The Q0001E (best estimate of annual flow, in cubic feet per second) and the DivDASqKM (divergence-routed cumulative area in square kilometers) from the EROM\_MA0001.dbf and the FCODE (feature code) from the NHD flowline shapefile are obtained for each flowing water body.
* The following FCODE values are used to filter the flowing water bodies: the 46000 (flowing water body), 46003 (intermittent), 46006 (perennial), and 46007 (ephemeral). This confines the analysis to streams and rivers rather than human-modified (canals, aqueducts, pipelines, other conveyances) or artificial paths (which could be both water bodies and NHD areas).
* The Q0001E parameter is converted to cubic meters per second to match the aquatic habitat bins.
* The ln-ln regression is generated for annual average flow and drainage area (**Table A 3-1.6**).
* Watershed drainage areas are then estimated for the 0.001 m3/s, 1 m3/s, and 100 m3/s flowing aquatic bins (**Table A 3-1.7**).

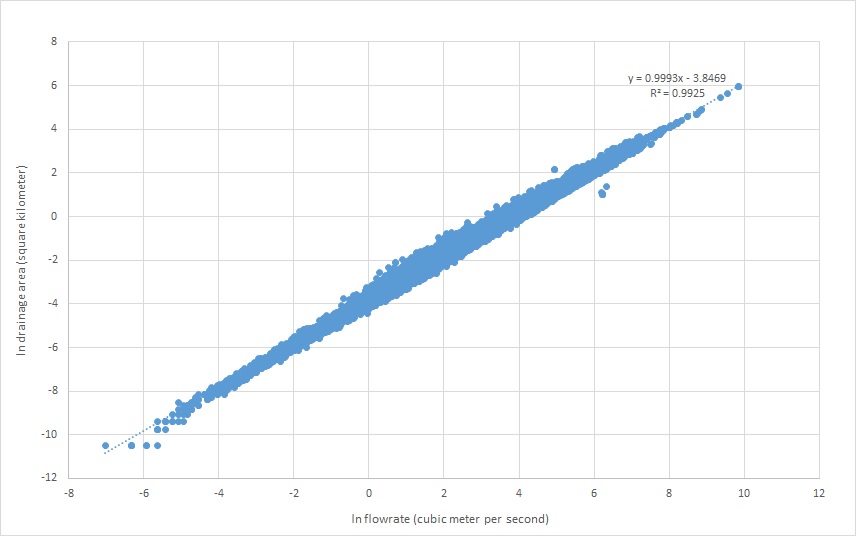


Figure A 3-1.4. Example regression plot of natural log(drainage area) to natural log(annual average flow rate) for flowing bins in HUC2 region 1.

**Table A 3-1.6**. **Regression statistics for the flowing bins by HUC2 regions**

| **HUC2 Region** | **Slope** | **Intercept** | **R2** |
| --- | --- | --- | --- |
| Region 01 | 0.999 | -3.847 | 0.993 |
| Region 02 | 1.026 | -4.256 | 0.984 |
| Region 03 | 1.016 | -4.302 | 0.975 |
| Region 04 | 1.004 | -4.478 | 0.968 |
| Region 05 | 0.973 | -4.046 | 0.982 |
| Region 06 | 1.020 | -4.001 | 0.985 |
| Region 07 | 0.834 | -3.713 | 0.955 |
| Region 08 | 1.038 | -4.385 | 0.997 |
| Region 09 | 0.872 | -5.380 | 0.862 |
| Region 10 | 0.884 | -5.153 | 0.742 |
| Region 11 | 0.658 | -3.480 | 0.621 |
| Region 12 | 0.651 | -3.381 | 0.809 |
| Region 13 | 0.690 | -6.209 | 0.649 |
| Region 14 | 0.917 | -6.726 | 0.513 |
| Region 15 | 1.067 | -7.837 | 0.866 |
| Region 16 | 0.620 | -5.263 | 0.559 |
| Region 17 | 1.066 | -5.347 | 0.595 |
| Region 18 | 0.880 | -5.141 | 0.443 |
| Region 191 | 1.066 | -5.347 | 0.595 |
| Region 20 | 0.903 | -2.846 | 0.874 |
| Region 21 | 1.023 | -3.745 | 0.964 |

1. Due to time constraints, HUC 19 values were derived from HUC 17 regression values.

##### Static Bins

Due to limitations in resolution of the NHDPlus v2 dataset (not fine enough resolution), a regression analysis for the static water body bins cannot be performed. For the static bins, the existing Mississippi (MS) corn scenario (determined to be a high runoff scenario) and all available weather stations throughout the country (242 stations) are used to estimate watershed areas (**Table A 3-1.8**). Runoff estimates from PRZM5/VVWM simulations, and the precipitation and evaporation data from the SAMSON weather input files are evaluated to determine the watershed size required to maintain the volume of the aquatic bin: 0.1 m3 (Bin 5), 10 m3 (Bin 6), and 20,000 m3 (Bin 7).

The conceptual approach and associated equations are provided in **Figure A 3-1.5**. Model simulations are grouped by HUC2 region, and watershed size is defined based on a median drainage area normalized to capacity (DANC) of 5-15 m2/m3. Using this approach, the static aquatic bin capacity is not exceeded, and the pesticide loading is conservative. More detailed calculations are provided in **SUPPLEMENTAL INFORMATION 1**.

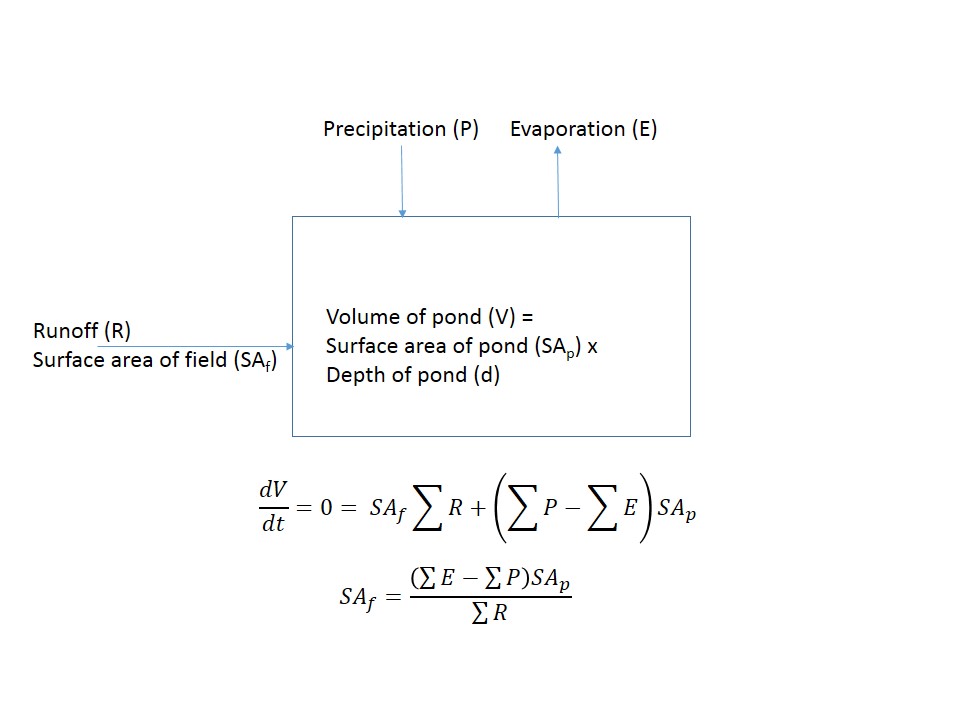


Figure A 3-1.5. Conceptual approach and equations for estimating watershed size for flowing Bins 5-7

For comparison, national DANCs from the USDA (1982) pond construction handbook range from ≤ 3.3 to ≥ 460 m2/m3. The USEPA OPP 2004 Scientific Advisory Panel (SAP) showed scenario-specific DANCs ranging from 3.3 to 260 m2/m3, for a constant field size (10 ha) and various water body volumes. Bin 7, similar to the Standard Pond, has DANCs ranging from 0.29 to 78.8 m2/m3 and a median watershed size across all HUC2 regions of 28.5 acres, compared to the Standard Pond watershed of 25 acres.



##### Estuarine and Marine

Current models are not designed to estimate environmental concentrations (EECs) in the estuarine and marine systems. Surrogate freshwater flowing or static bins instead are assigned to evaluate exposure in estuary and marine bins. Aquatic Bin 5 is used as a surrogate for pesticide exposure to species in tidal pools (Bin 8); aquatic Bins 2 and 3 are used for exposure to species at low and high tide (Bin 8), and aquatic Bins 4 and 7 are used for exposure to marine species that occasionally inhabit offshore areas (Bin 9). Likewise, the watershed areas estimated for the surrogate flowing and static bins are carried through in the estuarine and marine bin analyses.

**Table A 3-1.7. Watershed sizes for flowing waterbodies (Bins 2-4)**

| **HUC 2** | **Bin 2** | | | **Bin 3** | | | **Bin 4** | | | **R-squared** |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **Watershed**  **m2 (A)** | **Water body area (m2)** | **DANC**  **(m2/m3)** | **Watershed**  **m2 (A)** | **Water body area (m2)** | **DANC**  **(m2/m3)** | **Watershed**  **m2 (A)** | **Water body area (m2)** | **DANC**  **(m2/m3)** |
| 1 | 4.67E+04 (12) | 4.32E+02 | 1081 | 4.70E+07 (1.16E+04) | 5.48E+04 | 857 | 4.71E+09 (1.16E+06) | 2.75E+06 | 858 | 0.993 |
| 2 | 7.54E+04 (19) | 5.49E+02 | 1373 | 6.33E+07 (1.56E+04) | 6.36E+04 | 994 | 5.63E+09 (1.39E+06) | 3.00E+06 | 938 | 0.984 |
| 3 | 7.69E+04 (19) | 5.55E+02 | 1386 | 6.90E+07 (1.71E+04) | 6.65E+04 | 1039 | 6.43E+09 (1.59E+06) | 3.21E+06 | 1002 | 0.975 |
| 4 | 8.90E+04 (22) | 5.97E+02 | 1491 | 8.63E+07 (2.13E+04) | 7.43E+04 | 1161 | 8.46E+09 (2.09E+06) | 3.68E+06 | 1149 | 0.968 |
| 5 | 5.29E+04 (13) | 4.60E+02 | 1149 | 6.38E+07 (1.58E+04) | 6.39E+04 | 999 | 7.24E+09 (1.79E+06) | 3.40E+06 | 1063 | 0.982 |
| 6 | 5.78E+04 (14) | 4.81E+02 | 1202 | 5.05E+07 (1.25E+04) | 5.69E+04 | 888 | 4.61E+09 (1.14E+06) | 2.72E+06 | 849 | 0.985 |
| 7 | 2.17E+04 (5) | 2.95E+02 | 737 | 8.57E+07 (2.12E+04) | 7.41E+04 | 1157 | 2.14E+10 (5.29E+06) | 5.85E+06 | 1829 | 0.955 |
| 8 | 8.80E+04 (22) | 5.93E+02 | 1483 | 6.83E+07 (1.69E+04) | 6.61E+04 | 1033 | 5.77E+09 (1.42E+06) | 3.04E+06 | 949 | 0.997 |
| 9 | 1.73E+05 (43) | 8.33E+02 | 2082 | 4.79E+08 (1.18E+05) | 1.75E+05 | 2737 | 9.44E+10 (2.33E+07) | 1.23E+07 | 3841 | 0.862 |
| 10 | 1.37E+05 (34) | 7.41E+02 | 1853 | 3.41E+08 (8.42E+04) | 1.48E+05 | 2308 | 6.25E+10 (1.54E+07) | 1.00E+07 | 3126 | 0.742 |
| 11 | 5.48E+03 (1) | 1.48E+02 | 370 | 1.98E+08 (4.89E+04) | 1.13E+05 | 1758 | 2.16E+11 (5.34E+07) | 1.86E+07 | 5814 | 0.621 |
| 12 | 4.45E+03 (1) | 1.33E+02 | 334 | 1.80E+08 (4.44E+04) | 1.07E+05 | 1677 | 2.12E+11 (5.23E+07) | 1.84E+07 | 5753 | 0.809 |
| 13 | 3.63E+05 (90) | 1.21E+03 | 3014 | 8.05E+09 (1.99E+06) | 7.18E+05 | 11218 | 2.04E+112 (5.04E+07) | 1.01E+08 | 2020 | 0.649 |
| 14 | 8.20E+05 (202) | 1.81E+03 | 4527 | 1.54E+09 (3.79E+05) | 3.14E+05 | 4899 | 2.33E+11 (5.77E+07) | 1.93E+07 | 6039 | 0.513 |
| 15 | 2.39E+06 (590) | 3.09E+03 | 7725 | 1.54E+09 (3.81E+05) | 3.14E+05 | 4909 | 1.15E+11 (2.85E+07) | 1.36E+07 | 4244 | 0.866 |
| 16 | 7.05E+04 (17) | 5.31E+02 | 1327 | 4.84E+09 (1.20E+06) | 5.57E+05 | 8697 | 3.23E+102 (7.98E+06) | 1.14E+08 | 283 | 0.559 |
| 17 | 2.31E+05 (57) | 9.62E+02 | 2405 | 1.51E+08 (3.72E+04) | 9.82E+04 | 1535 | 1.13E+10 (2.80E+06) | 4.26E+06 | 1330 | 0.595 |
| 18 | 1.34E+05 (33) | 7.33E+02 | 1832 | 3.44E+08 (8.50E+04) | 1.48E+05 | 2319 | 6.45E+10 (1.59E+07) | 1.02E+07 | 3174 | 0.443 |
| 191 | 2.31E+05 (57) | 9.62E+02 | 2405 | 1.51E+08 (3.72E+04) | 9.82E+04 | 1535 | 1.13E+10 (2.80E+06) | 4.26E+06 | 1330 | 0.595 |
| 20 | 1.12E+04 (3) | 2.12E+02 | 529 | 2.34E+07 (5.77E+03) | 3.87E+04 | 604 | 3.82E+09 (9.45E+05) | 2.47E+06 | 773 | 0.874 |
| 21 | 4.55E+04 (11) | 4.27E+02 | 1067 | 3.88E+07 (9.59E+03) | 4.99E+04 | 779 | 3.50E+09 (8.63E+05) | 2.36E+06 | 739 | 0.964 |

1. Due to time constraints, HUC 19 values were derived from HUC 17 regression values.

2. Watershed sizes developed for Bin 4 using regressions developed in Section 3.1.3.3.1 were beyond watershed sizes noted for these regions. So watershed sizes were capped for these regions at the largest observed watershed area from NHDPlus.

**Table A 3-1.8. Watershed sizes for static waterbodies (Bins 5-7)**

| **HUC 2** | **Bin 5** | | **Bin 6** | | **Bin 7** | |
| --- | --- | --- | --- | --- | --- | --- |
| **Watershed**  **m2 (A)1** | **DANC**  **(m2/m3)** | **Watershed**  **m2 (A)1** | **DANC**  **(m2/m3)** | **Watershed**  **m2 (A)1** | **DANC**  **(m2/m3)** |
| 1 | 0.69 (1.70E-04) | 6.88 | 1.05E+02 (0.026) | 1.05 | 1.04E+04 (2.56) | 0.52 |
| 2 | 0.65 (1.60E-04) | 6.48 | 1.70E+02 (0.042) | 1.70 | 1.71E+04 (4.23) | 0.86 |
| 3 | 0.32 (8.02E-05) | 3.25 | 1.50E+02 (0.037) | 1.50 | 1.49E+04 (3.67) | 0.74 |
| 4 | 1.08 (2.66E-04) | 10.8 | 2.67E+02 (0.066) | 2.67 | 2.67E+04 (6.60) | 1.34 |
| 5 | 0.82 (2.03E-04) | 8.22 | 1.98E+02 (0.049) | 1.98 | 2.00E+04 (4.94) | 1.00 |
| 6 | 0.14 (3.45E-05) | 1.40 | 5.67E+01 (0.014) | 0.57 | 5.75E+03 (1.42) | 0.29 |
| 7 | 1.89 (4.66E-04) | 18.9 | 3.28E+02 (0.081) | 3.28 | 3.29E+04 (8.12) | 1.64 |
| 8 | 0.08 (2.07E-05) | 0.84 | 6.47E+01 (0.016) | 0.65 | 4.86E+03 (1.20) | 0.25 |
| 9 | 4.73 (1.17E-03) | 47.4 | 1.09E+03 (0.270) | 11 | 1.09E+05 (27) | 5.47 |
| 10 | 2.84 (7.01E-04) | 28.4 | 3.72E+03 (0.920) | 37 | 3.72E+05 (92) | 18.6 |
| 11 | 1.49 (3.68E-04) | 14.9 | 2.59E+03 (0.640) | 26 | 2.59E+05 (64) | 13.0 |
| 12 | 1.31 (3.23E-04) | 13.1 | 1.21E+03 (0.300) | 12 | 1.21E+05 (30) | 6.07 |
| 13 | 125.45 (3.10E-02) | 1,255 | 2.04E+04 (5.050) | 204 | 1.58E+06 (390) | 78.8 |
| 14 | 52.20 (1.29E-02) | 522 | 1.37E+04 (3.390) | 137 | 1.37E+06 (339) | 68.6 |
| 15 | 52.20 (1.29E-02) | 522 | 2.00E+04 (4.940) | 200 | 8.50E+05 (210) | 42.5 |
| 16 | 54.63 (1.35E-02) | 547 | 1.45E+04 (3.580) | 144 | 1.44E+06 (356) | 72.1 |
| 17 | 1.01 (2.50E-04) | 10.1 | 6.88E+03 (1.700) | 69 | 6.88E+05 (170) | 34.4 |
| 18 | 10.68 (2.64E-03) | 107 | 3.97E+03 (0.980) | 40 | 3.97E+05 (98) | 19.8 |
| 19 | 6.07 (1.05E-03) | 61 | 2.22E+03 (0.549) | 22 | 2.22E+05 (55) | 11.1 |
| 20 | 5.10 (1.26E-03) | 51 | 1.13E+03 (0.280) | 11 | 1.13E+05 (28) | 5.7 |
| 21 | 1.16 (2.87E-04) | 12 | 1.17E+02 (0.029) | 1.16 | 1.17E+04 (2.90) | 0.58 |

1 Based on 10th-25th% for Bin 5 to achieve a median DANC around 15 m2/m3, and 75th% for Bin 6 and Bin 7, to achieve a median DANC around 12 and 5 m2/m3, respectively.

#### Application Date Selection

In selecting application dates for aquatic modeling, a number of 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[[12]](#footnote-12), agricultural extension bulletins, and available state-specific use information.

Weather information is 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. If pest pressure or agronomic practice information is available to restrict the application period, then the wettest month during this period will be selected. 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 in decreasing average daily precipitation is provided in **Table A 3-1.9**.

**Table A 3-1.9. 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 |



#### Spray Drift Exposure

AgDRIFT v 2.1.1 (Spray Drift Task Force, 2011)[[13]](#footnote-13) is used to evaluate the deposition fractions for aerial, ground, and orchard applications based on label specifications. These fractions are then used in PRZM5/VVWM model simulations to capture 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 are provided in **Table A 3-1.10**. If more refined analysis is required, chemical-specific values will be derived and incorporated into a spray drift appendix.

**Table A 3-1.10.Default spray drift fractions for use in PRZM5/VVWM**

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Application Method** | **Drop size Distribution / Category** | **Spray drift fraction (unitless)1** | | | | | |
| **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| Aerial | Very fine to fine | 0.472 | 0.414 | 0.291 | 0.486 | 0.401 | 0.195 |
| **Fine to medium (default)** | 0.437 | 0.320 | 0.167 | 0.469 | 0.297 | 0.093 |
| Medium to coarse | 0.424 | 0.284 | 0.123 | 0.462 | 0.257 | 0.063 |
| Coarse to very coarse | 0.412 | 0.261 | 0.097 | 0.456 | 0.233 | 0.047 |
| Ground high boom | **Very fine to fine (default)** | 0.620 | 0.294 | 0.089 | 0.778 | 0.252 | 0.042 |
| Fine to medium/coarse | 0.215 | 0.079 | 0.024 | 0.336 | 0.067 | 0.012 |
| Ground low boom | Very fine to fine | 0.365 | 0.140 | 0.039 | 0.528 | 0.118 | 0.019 |
| Fine to medium/coarse | 0.154 | 0.054 | 0.016 | 0.251 | 0.045 | 0.008 |
| Airblast | **Sparse (default)** | 0.372 | 0.219 | 0.064 | 0.418 | 0.192 | 0.027 |
| Normal | 0.007 | 0.004 | 0.002 | 0.008 | 0.004 | 0.001 |
| Dense | 0.094 | 0.060 | 0.021 | 0.104 | 0.054 | 0.010 |
| Vineyard | 0.025 | 0.013 | 0.004 | 0.030 | 0.011 | 0.002 |
| Orchard | 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 2 – 2 m, Bin 3 – 8 m, Bin 4 – 40 m, Bin 5 – 1 m, Bin 6 – 10 m, and Bin 7 – 100 m.



### Issues Modeling Medium- and High-Flowing Waterbodies

#### Overview of Issues

Preliminary PRZM5/VVWM modeling of ESA aquatic Bins 3 (medium flow) and 4 (high flow) resulted in peak concentrations greater than several parts per million for all of the HUC2 regions and scenarios (*e.g.*, for chlorpyrifos 44,000-121,000,000 µg/L, 30 to 86,000 times higher than the solubility limit of chlorpyrifos, 1,400 µg/L). Several assumptions and model limitations have been identified as contributing to these overestimates:

* The large ratio of the watershed drainage area to the water body capacity.
* The entire watershed is treated with pesticide on the same application days (*e.g.*, when a runoff event occurs, the pesticide loading is from the entire watershed).
* Flow contributions from groundwater or sources upstream are not accounted for, and only runoff influences daily flows.
* Bins 3 and 4 are much larger water bodies than the PRZM5/VVWM Standard Pond and Index Reservoir.
* The instantaneous peak concentrations may not adequately reflect the pesticide concentrations in medium and high flowing water bodies (residence times << 1 day) within large watersheds. Though instantaneous peak concentrations have been used for smaller watersheds with water body residence times of > 1 day (*e.g.*, Standard Pond, Index Reservoir), larger watersheds may have runoff and pesticide loadings reaching the water body at varying times based on their upstream origin.

Several modifications are examined to address these issues and limitations in the ESA aquatic modeling approach for Bins 3 and 4. Atrazine monitoring data[[14]](#footnote-14) are also used in a case study to evaluate the recommended refinements, given the fit-for-purpose daily sampling. The atrazine monitoring data do not account for specifically where and when applications occur, as outlined in the NAS report, so some uncertainty may exist around whether the measured concentrations capture the maximum potential concentration that could occur along the length of the receiving stream rather than at the measured location. However, as discussed in **Section 3.1.4.4**, the monitoring data do reflect the integration of processes that are occurring within that watershed (*i.e.*, multiple applications, spread out over weeks, over a variety of landscapes), which is important when considering the estimation of aquatic concentrations in Bins 3 and 4. Additionally, there is a high degree of confidence in the data set due to daily sampling at a temporal and spatial scale consistent with the highest use of atrazine on corn.

The PRZM5/VVWM models are edge-of-field models, designed to assess pesticide and water runoff from treated fields into water bodies much smaller than those being assessed for the Bin 3 and 4 modeling. For the pilot draft BEs, EPA and the Services agreed to use existing tools for aquatic modeling as opposed to developing new models. For this reason, a watershed/basin-scale model is not incorporated for the pilot draft BEs. The techniques discussed below for estimating exposure in Bins 3 and 4 are being considered as interim approaches for improving pesticide concentration estimates in these flowing water bodies. EPA is currently developing a basin scale model, Spatial Aquatic Model (SAM), which may provide better estimates for Bins 3 and 4. However, this model is not far enough along in its development for use in these assessments. The methods explored may be an oversimplification of hydrologic process in the environment, but afford a means to estimate conservative pesticide concentrations in vulnerable water bodies. Recommendations for improving upon modeling these waterbodies from stakeholders, the scientific community, and the public are welcome and encouraged.



#### Modifications to Modeling Approach

Existing aquatic exposure modeling tools (PRZM5/VVWM) were used to evaluate aquatic EECs in the BEs, rather than models still under development such as the SAM. The SAM uses high resolution spatial data capturing the diversity of soil, land cover, weather, hydrology, and crop conditions/management practices and key fate and transport processes to estimate daily pesticide exposure concentrations in U.S. streams, rivers, lakes, and reservoirs.

The SAM is based on a new conceptual approach for exposure assessment, and will address many of the limitations identified in the modeling of medium and high-flowing water bodies (Bins 3 and 4). Some of the improvements developed as part of the SAM are examined further below. It should be noted that the SAM conceptual approach and its methods have not be reviewed by the FWS and NMFS, so the methods discussed below are considered preliminary investigations by the USEPA into methods that could improve Bin 3 and 4 EECs.

In the initial alpha version of SAM, like PRZM5/VVWM, pesticide loadings and runoff are assumed to enter receiving water bodies instantaneously. Water bodies are assumed to be completely mixed, with steady flow throughout the day. First-order washout occurs, and a mean daily concentration is calculated based on the pesticide loading, water body volume, and flow rate. Subsequent versions of SAM, however, account for longitudinal dispersion (concentration peak spreading and flattening) and the temporal re-distribution of pesticide concentration pulses at the pour point or receiving water body, based on in-stream travel times.[[15]](#footnote-15) In the ESA modeling approach, flowing water bodies (*i.e.*, Bins 2, 3 and 4) should similarly account for these processes.

As a contaminant moves downstream by advection and dispersion, the magnitude and distribution of its concentration changes. As seen in **Figure A 3-1.6,** a contaminant pulse tends to dampen and elongate as it moves from a headwater to successively larger water bodies. This can be represented by a pesticide load moving from a headwater stream (Strahler 1st order) such as Bin 2, to a higher order stream (3rd or 4th order) such as Bin 3, to a larger main stem river such as Bin 4. Smaller waterbodies, equivalent to Bin 2, are clustered around the headwater fringes of a watershed, while Bin 3 represents the habitat in the mid-reaches of the watershed and Bin 4 at the pour point.

Regardless of how the model is parameterized and the magnitude of the predicted exposures, the relationship between the exposures from Bin 2 to Bin 3 to Bin 4 must reflect what is expected in the environment based on watershed hydrology and contaminant transport. Initial modeling for the three pilot chemicals has demonstrated that, contrary to known contaminant transport processes, estimates have increased in magnitude and retained a sharp-pulsed shape. As a result, the following sections discuss alternatives explored to simulate these relationships.



**Figure A 3-1.6. Dispersion characteristics for Craigieburn, March 10, 1972[[16]](#footnote-16)**

##### Modifications Considered But Not Incorporated

Several modifications to the modeling of Bins 3 and 4 are explored and described below, but due to various reasons discussed are not further incorporated into the aquatic modeling for ESA.

###### Incorporation of Base Flow

PRZM5/VVWM allows for the adjustment of the average annual flow rate through the addition of base flow. Base flow (also referred to as drought flow, groundwater recession flow, low flow, low-water flow, low-water discharge and sustained or fair-weather runoff) is the portion of streamflow that comes from the sum of deep subsurface flow and delayed shallow subsurface flow[[17]](#footnote-17). The contribution of base flow to the total discharge of a flowing water body is evaluated here using the hydrograph separation method. A Python script was developed to apply a one-parameter digital filter separation method to every USGS stream gage in the country with discharge measurements (n = 16,632). A base flow index (BFI) (*i.e.*, ratio of annual base flow to the sum of annual base flow and annual runoff flow rates, with 0 being all runoff and 1 being all base flow) is developed for each gage station and HUC2 region (**Figure A 3-1.7**). The analysis indicates little deviation between the regional median BFIs, but substantial deviation within a HUC2 region. An interpolation (kriging) routine is applied to the index calculations to generate a national spatial distribution (**Figure A 3-1.8**). The results are then compared to the BFI developed by the USGS (**Figure A 3-1.9**) and the results appear very similar spatially. From this analysis, a mean BFI for any catchment can be developed.

Subsequent analysis shows whether a relationship exists between the magnitude of the flow rate for a stream and base flow, or more specifically, if streams with higher flow rates are more inclined to have higher base flows than those with lower flow rates. The theory is that streams with low flow rates are more influenced by watershed runoff than larger streams whose total flow may be influenced by both watershed runoff and base flow. To date, such an analysis has proven complex; therefore, simpler analyses of BFIs to drainage area are conducted. As presented in **Figure A 3-1.10**, the results do not appear to show a trend, so there is little evidence that the flow rates in Bins 3 and 4 are more influenced by base flow than smaller streams.

To evaluate the impacts of base flow addition on EECs (*i.e.*, impact of dilution flow), base flow rates are incorporated into a series of PRZM5/VVWM simulations, by including an assumed base flow of 50% of the annual average flow rate for the modeled aquatic bin (*e.g.*, 0.5 m3/s and 50 m3/s for Bins 3 and 4, respectively). While the addition of base flow reduces aquatic EECs for the various averaging periods (*e.g.*, daily, 4-day, 21-day, and 60-day average), it has no impact on the instantaneous peak EECs (**Table A 3-1.14**, comparison of Runs 1 and 2) because pesticide loading from the entire watershed to the water body is instantaneous, and daily peak EECs are calculated based on the volume of the water body alone (excluding base flow additions).

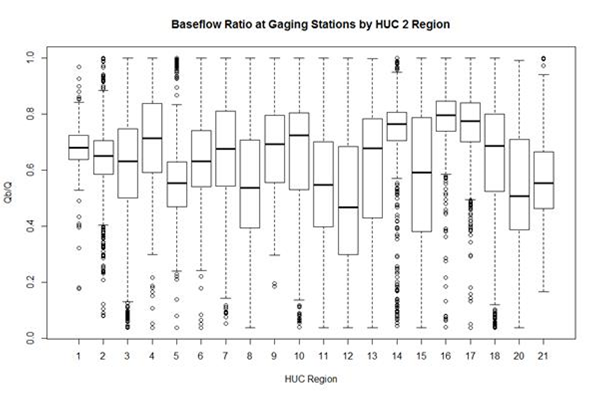


Figure A 3-1.7. EPA base flow analysis

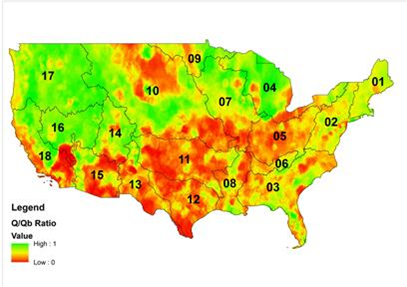


Figure A 3-1.8. Spatial representation of EPA base flow analysis across United States

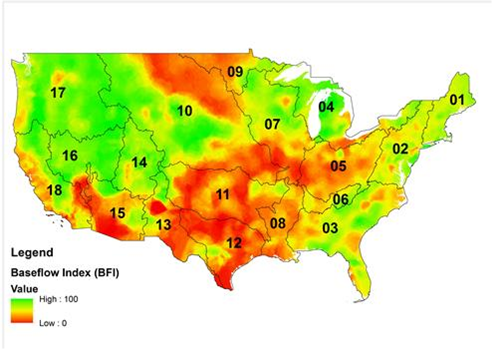


Figure A 3-1.9. USGS base flow index

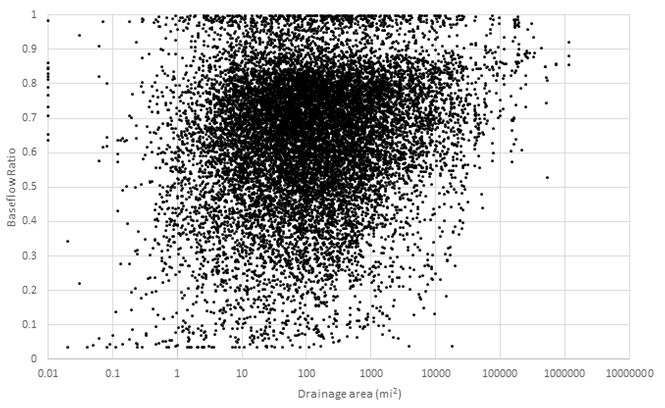


Figure A 3-1.10. Base flow versus drainage area



###### Percent Use Area and Percent Use Treatment Adjustment Factors

Percent cropped area (PCA) adjustment factors are used traditionally to account for the fraction of the watershed being planted with a particular crop and thus, being treated with pesticide. Additionally, USEPA OPP Health Effects Division (HED) and Biological Economics and Analysis Division (BEAD) have used percent cropped treated (PCT) values to further refine the fraction representing the planted crop area treated with pesticide in a watershed. The use of percent use area (PUA) factors, similar to PCA values, and adjustment factors accounting for more than just agricultural crop areas, have been proposed to adjust the watershed size treated with pesticide. However, the current ecological modeling approach does not evaluate the aggregate effect of all uses in a watershed (*e.g.*, combined impact of corn and nursery runoff to a water body). Combinations of PUAs (*e.g.*, not just corn, but corn and nursery use) would need to be incorporated to properly apply, as has been done in the past, PCAs. Additionally, percent use treatment (PUT) values obtained from BEAD could be evaluated to determine what fraction of a use actually is treated with the pesticide. A limitation of this approach is that PUT values are typically aggregated at the State level and do not reflect regional/local variations in areas treated (*e.g.*, regional/local pest pressures). As a result, there would be an additional level of uncertainty in PUT values, particularly for smaller watersheds. While the incorporation of these factors has some merit in refining EECs, the process needs to be evaluated further.



###### Adjustment of Water Body Length

Currently, the length of the water body is assumed to be the length of the watershed. However, this assumes the water body as a straight line down the center of the watershed, which splits the “square” watershed into two rectangles. In reality, the water body may meander through a watershed, or be a composite stream network, which would increase the effective length and effective volume of the water body, reducing the DANC. GIS tools could be used to evaluate representative watersheds throughout the country to determine the effective length of Bin 3 and 4 networks. Although this evaluation could not be conducted in the given time frame, it may be considered in future modeling efforts.

###### Spreading Out Applications

Currently, PRZM5/VVWM assumes applications occur on fixed dates, specified by the user as absolute dates (*e.g.*, February 12th) or dates relative to crop emergence. Usage data obtained from California’s Pesticide Use Reporting system and other pesticide use surveys show that for large watersheds not all farmers apply a pesticide on the same date, every day for 30 years. Applications in a watershed would be spread out over a period of time, based on pest pressure, relative planting date, precipitation events, and availability of the applicator. To account for this process, spreading out an application event over a fixed number of days has been explored. Retreatment intervals, may also affect the days over which applications can be made. Some preliminary modeling has been conducted with diazinon to evaluate the impact of spreading out the applications over time. A single application was spread uniformly, using the retreatment interval as the fixed window of application. Instantaneous peak EECs are only slightly reduced. Further analysis indicates that the fixed window is not long enough to allow for significant degradation to occur in the soil between applications. Thus, the loading to the water body is the same as if the application occurred on a single day. Additional evaluation of this approach is needed to identify the appropriate application window, annual maximum application rate, and retreatment intervals for diazinon.

The SAM has shown improved correspondence in the timing and magnitude of modeled and measured concentrations, by varying application dates each year according to crop growth stages, and spreading out applications over multiple days. Crop growth stages (e.g., planting, harvest) vary spatially and temporally, but only limited empirical information is available to define the growth stages for all crops nationally. Preliminary evaluations of this approach for atrazine applied to corn were supported by the Scientific Advisory Panel, and the Panel additionally recommended the use of other empirical data (e.g. crop insurance) when USDA Crop Progress reports are unavailable for a specific crop.[[18]](#footnote-18)

##### Modifications Explored and Incorporated into Modeling

The following changes were explored and incorporated into the case study at the end of this section.

###### Curve Number Adjustment

Curve numbers are selected for use in PRZM5/VVWM based on soil hydrologic group, land use, agricultural practices, and the hydrologic condition of the soil. Traditionally, the combination of soil, land cover, cropping, and management practices (e.g. conventional tillage, no-till) for particularly vulnerable sites results in high runoff curve numbers. While assuming a high runoff curve number is acceptable when modeling a representative field, the use of the same curve number for an entire watershed is a simplification, given the variety of land uses, hydrological conditions, and hydrologic soil groups within a watershed. A more refined approach would be to model fields and untreated areas within a watershed separately, each with an appropriate curve number. Although PRZM5/VVWM is not designed for this level of specificity, SAM is being built to capture this heterogeneity in runoff and field conditions. Since SAM is not yet available at the national scale, PRZM5/VVWM is modified to use a curve number based on the weighted-average of land uses and conditions within a given watershed. While runoff does not scale linearly with curve number, the recalculated curve numbers allow for the sensitivity of the estimated concentrations to be explored.

Curve number statistics are derived from two data sources: the USDA CDL for 2014 and the most recent gridded SSURGO (gSSURGO) product from USDA NRCS (February 2015). The gSSURGO product (native resolution 10 m) is resampled to a 30 m resolution to match the resolution of the CDL. Hydrologic soil group values from gSSURGO represent the major soil group of a given soil map unit. Curve numbers are generated for each 30 m pixel using methods established by the USDA NRCS[[19]](#footnote-19) .

Land cover classes from the CDL are matched with land cover types listed within the USDA guidance, consistent with the approach used in SAM[[20]](#footnote-20). Curve numbers for "fair" conditions are selected for each hydrologic unit code 12 (HUC12) catchment, based on the USGS Watershed Boundary Dataset[[21]](#footnote-21).

The 90th percentile and mean area-weighted HUC12 curve numbers are estimated for each HUC2 region and used for modeling Bins 3 and 4, respectively. As the watershed size associated with Bin 3 is comparable to a HUC12 catchment, the 90th percentile value of the HUC12 curve numbers is considered protective for modeling a HUC2 region. For Hawaii and Puerto Rico, curve number information is not available, so the highest 90th percentile and mean area-weighted curve numbers in the conterminous United States are assumed. The curve number values used for modeling are included in **Table A 3-1.11**.

**Table A 3-1.11. Curve number analysis**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **HUC2** | **N** | **Stdev** | **Bin 3** | **Bin 4** | **HUC2** | **N** | **Stdev** | **Bin 3** | **Bin 4** |
| 1 | 1,794 | 11 | 79 | 69 | 11 | 5,550 | 11 | 81 | 71 |
| 2 | 3,055 | 12 | 78 | 65 | 12 | 3,918 | 10 | 81 | 72 |
| 3 | 5,558 | 12 | 76 | 62 | 13 | 2,864 | 12 | 77 | 67 |
| 4 | 3,435 | 16 | 82 | 65 | 14 | 2,507 | 12 | 77 | 67 |
| 5 | 4,890 | 10 | 81 | 69 | 15 | 3,037 | 11 | 77 | 68 |
| 6 | 950 | 11 | 74 | 61 | 16 | 2,645 | 10 | 78 | 70 |
| 7 | 4,933 | 12 | 81 | 71 | 17 | 6,014 | 10 | 77 | 66 |
| 8 | 2,242 | 8 | 83 | 74 | 18 | 3,280 | 12 | 80 | 68 |
| 9 | 1,286 | 13 | 80 | 69 | 20\* | -- | -- | 83 | 74 |
| 10 | 10,474 | 10 | 81 | 74 | 21\* | -- | -- | 83 | 74 |

\* Information for the development of area-weighted curve numbers for HUC2 regions 20 and 21 was not available. The maximum 90th percentile (Bin 3) and mean (Bin 4) area-weighted curve numbers were selected and used for modeling purposes. This analysis was conducted prior to the development of HUC 19 scenarios, so HUC 19 is not included in this analysis.

###### Daily Flow Averaging

For flowing water bodies in the VVWM, the water body flow rate is traditionally estimated using the 30-year annual average flow estimated at the completion of the PRZM5 simulation. Average daily flow rates, however, can also be calculated in SAM, allowing for more accurate accounting of daily pesticide mass transport into and through a water body via runoff. The ESA modeling was modified to incorporate daily flow rates.

###### Adjustment of Water Body Dimensions

For flowing water bodies, water body volume can be estimated as the reach cross-sectional area multiplied by a representative length. In SAM, a representative length of 40 meters is used for flowing water bodies, based on typical longitudinal dispersion measured in rivers (Fischer, 1979[[22]](#footnote-22); Rutherford, 1994[[23]](#footnote-23)). The size of the watersheds modeled for ESA Bins 3 and 4 are approximately the same order of magnitude as the watershed sizes used in SAM for model evaluation. Daily concentrations are calculated for the “pour point” of each watershed catchment (**Figure A 3-1.11**), assuming runoff and pesticide mass are delivered to the outlet of the watershed. Regardless of the water body type (*e.g.*, stream, river, lake, or reservoir), concentrations are estimated using a completely mixed stirred tank reactor (CSTR) approach with an assumed water body volume and flow rate. In ESA exposure modeling, the water body dimensions for Bins 3 and 4 can be updated based on the widths provided by the Services (8 m for Bin 3 and 40 m for Bin 4). The area of the water body can be assumed as 320 m2 and 1,600 m2 for Bins 3 and 4, respectively, resulting in water body volumes of 320 m3 and 3,200 m3 for Bins 3 and 4, respectively.

****

**Figure A 3-1.11. Depiction of watershed network and water body/pour point (in red)**

###### Use of Daily Average EEC

The ESA modeling for flowing water bodies was also modified to use daily average concentrations, instead of traditional instantaneous peak values. Unlike the PRZM5/VVWM standard agricultural field (10 ha) or drinking water watershed (172.8 ha, an order of magnitude smaller than the smallest watershed for Bin 3), the watersheds associated with Bins 3 and 4 are larger and the pesticide mass transported by runoff to the water body would be distributed in time (not instantaneous). From an exposure perspective, daily EECs are also more meaningful to compare to toxicity studies with a minimum 2-day duration.

#### Modifications Evaluation, Case Study and Results

To assess the impact of the proposed modifications on modeling estimates (*e.g.*, area-weighted curve number, daily flow averaging, water body dimensions, and daily average concentration), model estimates for atrazine use on corn are compared to the Atrazine Ecological Exposure Monitoring Program (AEEMP) surface water samples collected from 2011 through 2014 (Bin 3). To assess concentrations in Bin 4, model results are compared to atrazine monitoring data for the Maumee River, OH.

Atrazine is one of the most studied pesticides with a fit-for-purpose monitoring dataset across a range of habitat types and vulnerabilities. Fit-for-purpose refers to the relationship between the monitored values and the source of exposures expected in that habitat. Fit-for-purpose is not meant to imply that the monitoring was coordinated with specific applications. For example, as the NAS noted field monitoring (*i.e.* edge of field) is most appropriate when evaluating a field scale model such as modeling Bin 2. [[24]](#footnote-24) When evaluating model performance at the watershed scale (*e.g.* Bins 3 and 4), the monitoring dataset should reflect watershed-level processes (*i.e.,* multiple applications spread over weeks, across a range of landscapes). Tools to evaluate monitoring data and how it relates to stream flow rates are being explored. Preliminary findings demonstrate that, for streams of the Bin 3 and 4 variety, specific information on chemical applications and releases may not be required (refer to **Section 4.2.3** for discussion of tools).

AEEMP data represents highly vulnerable 3rd and 4th order streams (Bins 3 and 4) and are reflective of higher end atrazine exposures in Midwestern corn and sorghum. The monitored watersheds have been selected using a runoff vulnerability approach that relied on atrazine use, soils, rainfall, and other parameters intended to represent upper bound exposure settings[[25]](#footnote-25). In 2003, the USGS Watershed Regression on Pesticides (WARP) model was used to identify those watersheds (at a HUC10 scale) with flowing water bodies that were predicted to be most vulnerable to atrazine surface water loading. The subgroup then identified 40 watersheds that gave a statistical representation of an upper tier of 1,172 most potentially vulnerable watersheds. Monitoring sites were located on flowing water bodies within these 40 watersheds. Over 10 years of monitoring has confirmed the monitored locations represent some of the most vulnerable settings in the corn and sorghum belt for atrazine. Since 2010, this conclusion has been reinforced by the tracking of individual grower applications in AEEMP watersheds with the highest exposures, collected to evaluate conditions and practices driving high exposures. By tracking individual applications and agronomic practices coupled with daily monitoring, there is high confidence that AEEMP monitoring reflects actual use in these settings. This information, along with the fact that these watersheds yield some of the highest exposures across all monitored sites, indicates that these data provide a direct measure of atrazine exposure appropriate for risk assessment purposes.

A subset (29) of the AEEMP sites, which had daily or near daily monitoring data from 2011 to 2014, was selected for an alternatives analysis. The subset of 29 sites draws from the original 40 sites that continually had high levels of atrazine in water, as well as 25 additional sites identified as vulnerable based on a revised set of watershed criteria described in the 2009 SAP (<http://www.regulations.gov/#!docketBrowser;rpp=25;po=0;dct=SR;D=EPA-HQ-OPP-2009-0104>). These 25 new sites are selected based on refined vulnerability characteristics that further maximized the potential for high atrazine concentrations. A summary of the monitoring results and the properties of the watersheds and water bodies are provided in **Table A 3-1.12**.

The Maumee River dataset is a collection of samples collected by the Heidelberg University National Center for Water Quality Research from 1983 to 1999. Years 1987 and 1994 have been selected, as these are the years with annual average flow rates closest to the target flow rate for Bin 4 of 100 m3/s. Maximum daily average atrazine concentrations ranged from 2 to 39 µg/L from 1983 to 1999, with a maximum average daily concentration of 10 and 4 µg/L for 1987 and 1994, respectively. While the sampling intervals varied from 2 to 30 days, sampling became more frequent (daily) from May to August.

Data provided by the registrant and derived from sales data, dealership interviews, registrant surveys, and farmer interviews indicate that applications occurred in sampled areas and were slightly lower than label maximum rates (i.e., generally around 1.5-2 lbs/acre/year instead of 2.5 lbs/acre/year). Atrazine applications to corn are most often pre-emergence (mid-April through mid-May in the major corn-growing areas). Post-emergence applications are most likely to occur up through the end of June, until corn reaches 12" in height. There is also some variability in timing based on geographical location. As a result, monitoring data were collected from approximately March to August. Applications to corn are modeled using a range of application dates from March 1 to August 15. Fate parameters used in aquatic modeling are provided in **Table A 3-1.13**. Monitored concentrations confirm that applications occurred during the predicted window, with little to no atrazine initially and increasing concentrations thereafter.

Modeling is conducted using watershed and water body sizes for the various HUC2 regions. Watershed areas representative of the monitoring datasets (HUC2 regions 5, 6, 7, 8, 10, 11, and 12) range from 5.05E+07 to 3.41E+08 m2 for Bin 3 and 4.61E+09 to 2.16E+11 m2 for Bin 4.

Modeling results for the refinements are presented in **Table A 3-1.14** for Bins 3 and 4 (Maumee HUC 4). The monitoring results presented in **Table A 3-1.12** have not been adjusted to account for a sampling interval that may have missed the peak. In most cases, the sampling was done on a daily or near daily basis, so an adjustment for these values would not be needed. Also, the modeled application rates are slightly higher than those reported by the registrant for the monitoring time periods. Therefore, modeling results may be lower than those reported in **Table A 3-1.14,** if the application rates are reduced.

As seen in **Table A 3-1.14**, the EECs decrease as the alternatives are progressively incorporated into the modeling runs. The best agreement with monitoring results occurs in Run 6, where a combination of the incorporation of daily flow averaging, a 50% base flow value, area-weighted curve number, water body adjustment to reflect a pour point evaluation, and the use of daily averages instead of instantaneous peaks is employed. Slightly less agreement occurs in Run 5 where the incorporation of base flow is removed. While the median BFI across the HUC 2s appears to hover around 50% (**Figure A 3-1.9**) and provides some justification for the use of a 50% base flow value at a HUC2 scale, there is no indication that this would be protective at a more local scale.

**Table A 3-1.12. Summary of AEEMP/Maumee monitoring results**

| **Site ID** | **Watershed Name** | **Watershed size (A)[m2]1** | **Year** | **Average Flow (cms)** | **Date Range** | **Number of Samples2** | **Maximum Concentration (ug/L)** |
| --- | --- | --- | --- | --- | --- | --- | --- |
| IA-03  (HUC 7) | Lick Creek | 21,855  [8.84E+07] | 2011  2012 | 1.97  0.29 | 4/1-8/23  3/23-8/1 | 145 (145)  132 (132) | 39.75  17.80 |
| IA-04  (HUC 7) | West Branch Sugar Creek | 15,321  [6.20E+07] | 2011  2012  2013  2014 | 0.63  0.31  0.44  0.35 | 4/1-8/2  3/23-8/1  3/31-8/13  4/13-8/18 | 124 (124)  132 (132)  136 (136)  103 (127) | 35.95  78.43  46.12  142.41 |
| IA-05  (HUC 7) | Little Sugar Creek | 7,029  [2.84E+07] | 2011  2012  2013  2014 | 0.12  0.05  0.09  0.04 | 4/1-8/2  3/23-8/1  3/31-8/13  4/13-8/18 | 124 (124)  132 (132)  136 (136)  100 (127) | 168.97  33.48  56.52  344.26 |
| IL-10  (HUC 7) | Felky Slough | 21,656  [8.76E+07] | 2011  2012 | 4.09  2.21 | 4/1-8/23  3/23-8/1 | 145 (145)  132 (132) | 11.45  5.42 |
| IL-12  (HUC 7) | Pigeon Creek | 19,298  [7.81E+07] | 2011  2012 | 2.71  0.64 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 58.86  60.73 |
| IL-14  (HUC 7) | Blue Grass Creek | 13,251  [53.6E+07] | 2011  2012 | 0.19  0.15 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 44.22  36.27 |
| IL-15  (HUC 7) | Elliott Creek | 9,539  [3.86E+07] | 2011  2012 | 1.00  0.32 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 50.24  71.85 |
| IL-16  (HUC 5) | Big Muddy Creek | 19,923  [8.06E+07] | 2011  2012 | 5.03  0.55 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 14.69  73.67 |
| IL-17  (HUC 7) | Limestone Creek | 8,781  [3.55E+07] | 2011  2012 | 2.73  1.14 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 228.18  39.36 |
| IN-12  (HUC 5) | Sixmile Creek | 19,547  [7.91E+07] | 2011  2012 | 2.40  0.69 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 34.80  10.66 |
| KS-01  (HUC 10) | Delaware River | 24,963  [1.01E+08] | 2011  2012 | 1.08  0.41 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 10.10  19.26 |
| KS-02  (HUC 10) | Mooney Creek | 6,664  [2.70E+07] | 2011  2012 | 0.97  0.13 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 24.13  10.58 |
| KS-03  (HUC 10) | Crooked Creek | 23,029  [9.32E+07] | 2011  2012 | 1.97  0.22 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 32.14  24.69 |
| LA-04  (HUC 8) | Branch of Boeuf River | 12,151  [4.92E+07] | 2011  2012  2013  2014 | 0.44  0.56  0.44  1.67 | 2/27-8/2  2/26-8/1  2/24-8/13  2/23-8/4 | 157 (157)  158 (158)  171 (171)  111 (162) | 167.44  58.79  193.65  76.82 |
| MO-01  (HUC 7) | South Fabius River | 7,439  [3.01E+07] | 2011  2012  2013  2014 | 0.44  0.18  0.75  0.24 | 4/1-8/3  3/23-8/1  3/31-8/13  4/13-8/4 | 125 (125)  132 (132)  136 (136)  99 (113) | 18.82  13.61  19.18  16.85 |
| MO-02  (HUC 7) | Youngs Creek | 18,023  [7.29E+07] | 2011  2012  2013  2014 | 0.03  0.02  0.05  0.78 | 4/1-8/2  3/23-8/1  3/31-8/13  4/13-8/4 | 124 (124)  132 (132)  136 (136)  99 (113) | 40.71  8.72  64.08  39.63 |
| MO-04A  (HUC 7) | South Fabius River | 5,382  [2.18E+07] | 2011  2012 | 0.19  0.02 | 4/1-8/3  3/23-8/1 | 125 (125)  132 (132) | 88.94  48.76 |
| MO-05  (HUC 7) | Long Branch | 16,192  [6.55E+07] | 2011  2012 | 0.25  0.30 | 4/1-8/23  3/23-8/1 | 145 (145)  132 (132) | 70.53  13.97 |
| MO-05B  (HUC 7) | Lower Long Branch | 116,781  [4.73E+08] | 2011  2012 | 3.99  3.44 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 21.56  25.99 |
| MO-06  (HUC 7) | White Cloud Creek | 24,130  [9.84E+07] | 2011  2012 | 0.95  2.17 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 102.43  47.96 |
| MO-07N  (HUC 10) | Honey Creek | 17,123  [6.93E+07] | 2011  2012  2013  2014 | 4.30  1.10  2.51  5.73 | 4/1-8/1  3/23-8/1  3/31-8/13  4/13-8/18 | 124 (124)  132 (132)  136 (136)  103 (127) | 90.87  41.13  42.4  285.86 |
| MO-08  (HUC 7) | West Fork Cuivre River | 24,942  [1.01E+08] | 2011  2012  2013  2014 | 0.95  0.64  0.79  0.68 | 4/1-8/2  3/23-8/1  3/31-8/13  4/13-8/18 | 125 (125)  132 (132)  136 (136)  103 (127) | 68.19  66.34  41.23  64.82 |
| NE-04  (HUC 10) | Big Blue River, Upper Gage | 13,852  [5.61E+07] | 2011  2012  2013  2014 | 0.18  0.10  0.50  0.02 | 4/1-8/2  3/23-8/1  3/31-8/13  4/13-8/4 | 124 (124)  132 (132)  136 (136)  76 (113) | 12.98  8.40  6.91  4.15 |
| NE-05  (HUC 10) | Muddy Creek | 31,540  [1.28E+08] | 2011  2012 | 0.59  0.22 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 58.71  9.11 |
| NE-09  (HUC 10) | Weeping Water Creek | 23,826  [9.64E+07] | 2011  2012 | 3.30  2.84 | 4/1-8/2  3/23-8/1 | 124 (124)  132 (132) | 4.77  17.18 |
| OH-05  (HUC 4) | Auglaize River | 20,159  [8.16E+07] | 2011  2012 | 2.04  0.46 | 4/1-8/9  3/23-8/1 | 131 (131)  132 (132) | 30.61  14.82 |
| OH-06  (HUC 5) | Toti Creek | 6,446  [2.61E+07] | 2011  2012 | 2.97  0.85 | 4/1-8/9  3/23-8/1 | 131 (131)  132 (132) | 47.15  51.23 |
| TX-01  (HUC12) | Chocolate Bayou | 23,723  [9.60E+07] | 2011  2012  2013  2014 | 0.09  0.10  0.07  0.05 | 12/1/10-8/2  1/15-6/29  2/10-7/2  12/3/13-7/10 | 245 (245)  167 (167)  143 (143)  114 (219) | 6.37  133.89  56.03  39.25 |
| TX-02  (HUC12) | Branch of Chiltipin Creek | 14,221  [5.75E+07] | 2011  2012 | NR  0.01 | 12/1/10-8/2  1/15-6/29 | 245 (245)  167 (167) | 0.08  9.85 |
| (HUC 4) | Maumee River | 4.02E+06  [1.63E+10] | 1987  1994 | 105  103 | 2/2-12/14  1/31-12/19 | 53 (315)  61 (350) | 9.92  4.02 |

1 Watershed areas provided by atrazine registrant.

2 Numbers in parenthesis represent the maximum number of daily samples that could have been collected between the range of sampling dates.

**Table A 3-1.13. PRZM5/VVWM modeling parameters**

|  |  |
| --- | --- |
| **Parameter (units)** | **Input Value** |
| Molecular Weight (g/mol) | 215.69 |
| Water Solubility (mg/L) 20 °C | 33 |
| Vapor Pressure (torr) 25 °C | 3x10-7 |
| Number of Applications | 2 |
| Application Rate (kg a.i./ha) | 2.24, 0.56 |
| First Application Date | 3/1-8/15 |
| Retreatment Interval (days) | 15 |
| Spray Drift | 0.12 (aerial) |
| Application Efficiency | 0.95 (aerial) |
| Crop Application Method | Foliar |
| Hydrolysis Half-life (days) | 0 |
| Aqueous Photolysis (days) | 168 |
| Aerobic Soil Metabolism Half-life (days) | 417 |
| Aerobic Aquatic Metabolism Half-life (days) | 277 |
| Anaerobic Aquatic Metabolism Half-life (days) | 589 |
| Koc (mL/goc) | 75 |

**Table A 3-1.14. Case study, atrazine PRZM5/VVWM modeling results\***

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Run | Watershed Area (m2), Water Body Volume (m3), CN | 1-in-10 Peak (µg/L) | Watershed Area (m2), Water Body Volume (m3), CN | 1-in-10 Peak (ug/L) | Watershed Area (m2), Water Body Volume (m2), CN | 1-in-10 Peak (ug/L) |
|  | HUC 5 (74 µg/L) | | HUC 6 | | HUC 7 (344 µg/L) | |
| 11 | 6.39E+7, 6.39E+4, 91 | 2,640-11,700 | 5.06E+7, 5.69E+4, 89 | 2,280-10,300 | 8.58E+7, 7.41E+4, 91 | 3,540-14,700 |
| 22 | 6.39E+7, 6.39E+4, 91 | 2,600-11,700 | 5.06E+7, 5.69E+4, 89 | 2,130-10,300 | 8.58E+7, 7.41E+4, 91 | 3,440-14,600 |
| 33 | 6.39E+7, 6.39E+4, 81 | 1,160-4,580 | 5.06E+7, 5.69E+4, 74 | 525-3,100 | 8.58E+7, 7.41E+4, 81 | 1,170-9,230 |
| 44 | 6.39E+7, 3.20E+2, 91 | 628-3,120 | 5.06E+7, 3.20E+2, 89 | 432-685 | 8.58E+7, 3.20E+2, 91 | 1,310-11,600 |
| 55 | 6.39E+7, 3.20E+2, 81 | 459-1,100 | 5.06E+7, 3.20E+2, 74 | 263-421 | 8.58E+7, 3.20E+2, 81 | 424-7,020 |
| 66 | 6.39E+7, 3.20E+2, 81 | 206-436 | 5.06E+7, 3.20E+2, 74 | 108-269 | 8.58E+7, 3.20E+2, 81 | 220-337 |
|  | HUC 8 (194 µg/L) | | HUC 10a (285 µg/L) | | HUC 10b (285 µg/L) | |
| 11 | 6.84E+7, 6.61E+4, 91 | 3,840-24,600 | 3.41E+8, 1.48E+5, 94 | 9,240-30,000 | 3.41E+8, 1.48E+5, 94 | 3,760-15,400 |
| 22 | 6.84E+7, 6.61E+4, 91 | 3,840-24,600 | 3.41E+8, 1.48E+5, 94 | 9,240-30,000 | 3.41E+8, 1.48E+5, 94 | 3,760-15,400 |
| 33 | 6.84E+7, 6.61E+4, 83 | 2,510-17,800 | 3.41E+8, 1.48E+5, 81 | 1,830-15,600 | 3.41E+8, 1.48E+5, 81 | 1,080-5,980 |
| 44 | 6.84E+7, 3.20E+2, 91 | 701-13,900 | 3.41E+8, 3.20E+2, 94 | 758-5,290 | 3.41E+8, 3.20E+2, 94 | 711-3,930 |
| 55 | 6.84E+7, 3.20E+2, 83 | 555-10,800 | 3.41E+8, 3.20E+2, 81 | 415-4,070 | 3.41E+8, 3.20E+2, 81 | 298-1,800 |
| 66 | 6.84E+7, 3.20E+2, 83 | 296-496 | 3.41E+8, 3.20E+2, 81 | 269-490 | 3.41E+8, 3.20E+2, 81 | 141-452 |
|  | HUC 11a | | HUC 11b | | HUC 12a (134 µg/L) | |
| 11 | 1.98e+8, 1.13e+5, 91 | 9,350-34,000 | 1.98e+8, 1.13e+5, 94 | 7,330-20,400 | 1.80e+8, 1.07e+5, 91 | 3,130-28,700 |
| 22 | 1.98e+8, 1.13e+5, 91 | 9,350-34,000 | 1.98e+8, 1.13e+5, 94 | 7,120-20,400 | 1.80e+8, 1.07e+5, 91 | 3,130-28,500 |
| 33 | 1.98e+8, 1.13e+5, 81 | 2,600-12,800 | 1.98e+8, 1.13e+5, 81 | 1,170-11,600 | 1.80e+8, 1.07e+5, 81 | 1,200-15,600 |
| 44 | 1.98e+8, 3.20e+2, 91 | 794-8,080 | 1.98e+8, 3.20e+2, 94 | 705-5,710 | 1.80e+8, 3.20e+2, 91 | 809-2,560 |
| 55 | 1.98e+8, 3.20e+2, 81 | 474-1,820 | 1.98e+8, 3.20e+2, 81 | 481-2,000 | 1.80e+8, 3.20e+2, 81 | 506-1,170 |
| 66 | 1.98e+8, 3.20e+2, 81 | 282-454 | 1.98e+8, 3.20e+2, 81 | 272-498 | 1.80e+8, 3.20e+2, 81 | 353-486 |
|  | HUC 12b (134 µg/L) | | Maumee (HUC 4) (9.92 µg/L) | | Standard Pond | |
| 11 | 1.80e+8, 1.07e+5, 91 | 4,760-29,300 | 8.46e+9, 7.36e+6, 92 | 4,490-12,700 | 1E+4, 1E+3, 91 | 165 |
| 22 | 1.80e+8, 1.07e+5, 91 | 4,760-29,300 | 8.46e+9, 7.36e+6, 92 | 4,490-12,200 |  |  |
| 33 | 1.80e+8, 1.07e+5, 81 | 1,990-13,600 | 8.46e+9, 7.36e+6, 65 | 111-1,890 |  |  |
| 44 | 1.80e+8, 3.20E+2, 91 | 755-1,780 | 8.46e+9, 3.20E+3, 92 | 688-3,660 |  |  |
| 55 | 1.80e+8, 3.20E+2, 81 | 446-856 | 8.46e+9, 3.20E+3, 65 | 154-1,470 |  |  |
| 66 | 1.80e+8, 3.20E+2, 81 | 272-486 | 8.46e+9, 3.20E+3, 65 | 64-180 |  |  |

\* Concentrations appearing next to the HUC number are the highest daily peak concentration reported from the AEEMP/Maumee monitoring dataset for that HUC (see Table 2).

1. Watershed and water body sizes determined via GIS, curve number is conservative estimate, instantaneous peak concentration.

2. Base flow of 1 and 100 m3/s were included in the run, instantaneous peak concentration.

3. Watershed and water body sizes determined via GIS, curve number estimated via area weighting, instantaneous peak concentration.

4. Watershed and water body sizes determined via GIS, curve number conservative estimate, daily flow averaging, daily average concentration.

5. Watershed and water body sizes determined via GIS, curve number estimated via area weighting, daily flow averaging, daily average concentration.

6. Watershed and water body sizes determined via GIS, curve number estimated via area weighting, 50% of targeted flow added as base flow, daily average concentration.

#### Modifications Evaluation, Pilot Chemicals, Draft BEs

The approaches presented in **Section 3.1.4.2** are applied to the modeling of the three ESA pilot chemicals (chlorpyrifos, diazinon, and malathion) to determine if similar impacts will result for chemicals with different chemical properties (solubility, sorption, degradation, etc.). The results show that in some instances, EECs similarly decrease (*e.g.*, they are lower than those as seen without the approaches) for atrazine, while in other cases the EECs increase several orders beyond the chemical’s solubility limit. Additionally, Bin 3 and 4 EECs are higher than those obtained for Bin 2 (*e.g.*, for chlorpyrifos, the maximum daily EECs for Bin 3 are 1 - 200 times higher than those for Bin 2 and the values for Bin 4 are 2 – 23,000 times higher than the Bin 2 estimates).

The current modeling approach using PRZM5/VVWM likely will not generate EECs for Bins 3 and 4 that are realistic or protective. As a result, the following qualitative approach is considered for use in assessing Bins 3 and 4.

Existing atrazine monitoring datasets are used to evaluate the relative magnitude in EECs as a chemical moves from lower to higher order streams/waterbody volumes as represented by Bins 2-4. Atrazine is one of the most studied pesticides with a fit-for-purpose set of monitoring data across a range of habitat types and vulnerabilities. Fit-for-purpose refers to the relationship between the monitored values and the source of exposures expected in that habitat. Fit-for-purpose is not meant to imply that the monitoring was coordinated with specific applications. For example, as the NAS noted field monitoring (*i.e.* edge of field monitoring) is most appropriate when evaluating a field scale model such as modeling for Bin 2. [[26]](#footnote-26) When considering a relevant monitoring data set for evaluating model performance at the watershed scale (*e.g.* Bins 3 and 4), the monitoring should reflect the watershed processes that are occurring. The watershed is in effect the integrator of all processes occurring within that watershed (*i.e.* multiple applications, spread out over weeks, over a variety of landscapes) and does not reflect any one single contaminant release.

A selection of data sources is used to qualitatively evaluate the levels of EECs anticipated for all three flowing water bins. The following sources of data have been used to represent the progression of exposures from most vulnerable headwater streams to major flowing rivers.

* Wauchope[[27]](#footnote-27) and other authors catalog a series of field monitoring studies that are consistent with the NAS description of field monitoring and are tied to atrazine applications. These data are used to represent Bin 2 estimates.
* AEEMP data, discussed above, present daily sampling from 3rd through 5th order streams in high intensity atrazine use areas. These data are used to qualitatively represent Bin 3 estimates.
* Heidelberg College monitoring data from the Maumee River in Ohio, discussed above, present daily samples from a 6th to 7th order stream. These data are used to qualitatively represent Bin 4 estimates.

The key concept to consider when making these comparisons is the relative magnitudes of exposures from the monitoring data and how Bin 3 and 4 EECs might be expected to change relative to the modeled Bin 2 estimates. For example,

* Edge of field monitoring from Wauchope shows that in the most vulnerable case (i.e. headwater stream adjacent to treated field) that atrazine typically occurs between 1 and 5 ppm.
* Conversely, the AEEMP data, which represents an integration of multiple applications across a mid-sized watershed, shows a spread of exposures over time for atrazine, typically between 200 and 400 ppb, representing a 5x decrease in peak exposures.
* Finally, the Maumee monitoring data typically ranges between 1 and 10 ppb (though up to 60 ppb was detected in years with less frequent sampling) representing an order of magnitude decrease in peak exposure from the edge of field setting to a larger flowing river.

An additional line-of-evidence for a decrease in pesticide concentration with increasing water body size and flow is to consider the relative impact of dilution that would occur to an instantaneous pesticide release into the different water bodies from the same adjacent field. Assuming the same pesticide mass loading from the field, the level of dilution, based on the volume of the water bodies, would be 40 times greater for Bin 3 as compared to Bin 2 [1 m x 8 m / (0.1 m x 2 m) = 40] and would be 10 times greater for Bin 4 as compared to bin 3 [2 m x 40 m / (1 m x 8 m) = 10]. As the flow rates for these water bodies also increases by orders of magnitude (0.001 m3/s for Bin 2 to 1 m3/s for Bin 3 to 100 m3/s for Bin 4), an increase due to advection-dispersion would also be expected, reducing the pesticide concentrations in the water bodies further. While this approach does not account for additional pesticide loadings from other sources upstream of the field, it does provide the relative reduction in loading potential one might expect at the edge of the field.

The monitoring data sets, when viewed together and in sequence from the headwater to the major river, show a clear pattern that matches the concept of downstream dispersion one would expect as a pesticide moves from Bin 2 to Bin 4 successively (**Figure A 3-1.6**). Coupled with the relative increase in dilution potential, based on the increase in volume and flowrate, as one moves from Bin 2 to Bin 4, these lines-of-evidence can be used to characterize the expected EECs in the various flowing bins. As such, an approach is being used where Bin 2 EECs are generated using the PRZM5/VVWM. Bin 3 EECs are characterized as being conservatively 5 and 10 times lower than the Bin 2 EECs, and the Bin 4 EECs are characterized as being conservatively 5 and 10 times lower than the Bin 3 EECs. Bin 2 EECs will also be provided for species that inhabit Bins 3 and 4 as a metric for exposure that could occur near shorelines where complete mixing has not yet occurred. It should be noted that, as a result of using this scaling technique, there is uncertainty around the Bin 3 and 4 EECs.



## Modifications Evaluation, Pilot Chemicals, Final BEs

A number of recommendations received during the public comment period which are discussed below, were incorporated into EFED’s methodology for estimating PRZM5/VVWM model generated Bin 3 and 4 EECs.

Specifically, daily (24-hour) mean concentrations have been adopted in place of the initial (time zero) concentrations that had been previously used as acute EECs in the draft BEs. From an exposure perspective, daily mean concentrations provide a more meaningful metric, than do initial concentrations, for comparison against the results of acute toxicity studies, where organisms are exposed to a pesticide for at least 48 hours. Additionally, the initial concentrations are essentially a hypothetical construct that is inherent in the way EFED currently calculates daily mean concentrations. When a modeled receiving water body is relatively large compared with its watershed and has a residence time (*i.e.*, the amount of time water spends in a waterbody) on the order of months, which is the case for EPA’s Index Reservoir, or is assumed to have no flow out of it, which is the case for EPA’s standard pond, daily mean and initial concentrations are typically very close to each other. However, as the size (volume) of receiving water body shrinks in comparison to its watershed, and particularly as residence times decline below one day, as is the case for the flowing bins used in the BEs, the initial and daily mean concentrations begin to diverge, with smaller residence times resulting in a larger divergence. This is a result of the assumptions inherent in the calculation methodology, and not a reflection of fundamental differences in physical or chemical processes at work in smaller receiving water bodies.

Another recommendation that has been adopted is incorporation of baseflow into the flow for Bins 3 and 4. Baseflow is the portion of streamflow that comes from subsurface discharge to a stream or river (as opposed to direct overland runoff). Baseflows for use in modeling were derived from regionally-representative estimates of baseflow index (BFI), defined as the fraction of total (long-term) stream flow that consists of baseflow. BFI values were extracted from EPA’s Stream-Catchment (StreamCat) dataset, which provides estimates of this property for the millions of flowing reaches across the country that are represented in the National Hydrography Dataset (NHD). Within each HUC 2, the average BFIs for reaches with flows similar to those of Bins 3 and 4 were calculated and tabulated (**Table A 3-1.15**). The HUC/Bin specific BFIs are currently applied to the annual average flowrate assigned for each bin (1 m3/s and 100 m3/s for Bin 3 and 4, respectively), in order to provide a HUC 2-specific baseflow value for use in each simulation.

For HUC 3, BFI data are available for three separate regions in NHDPlus: South Atlantic North (3N), South Atlantic South (3S), and South Atlantic West (3W). As the aquatic modeling for HUC 3 is not subdivided into a and b categories to account for changes in precipitation, the region with the lowest median BFI, 3N, was used, as it would generate lower baseflow estimates and more protective EECs. Similarly, for HUC 10, data are available for two separate regions in NHDPlus: the Upper Missouri (10U) and Lower Missouri (10L). As the aquatic modeling for HUC 10 is subdivided into a and b categories to account for changes in precipitation, a simplified map analysis was used to assign the BFI for 10L to modeling for HUC 10a and the BFI for 10U to modeling for HUC 10b. Lastly, it should be noted that BFI data are not available for HUCs 19 (AK), 20 (HI), and 21 (PR). As such, BFI data for nearby surrogate regions with similar meteorological conditions (HUC 17 for HUCs 19 and HUC 20 and HUC 3 for HUC 21) were used until such time as data become available for these regions.

**Table A 3-1.15 Baseflow Index Statistics for Each HUC 2**

| HUC | Count | Mean | Std  Dev | Min | Percentile | | | Max | HUC | Count | Mean | Std  Dev | Min | Percentile | | | Max |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| 25% | 50% | 75% | 25% | 50% | 75% |
| 1 | 65368 | 51 | 4 | 40 | 48 | 50 | 54 | 72 | 10a | 65968 | 52 | 14 | 17 | 41 | 51 | 61 | 86 |
| 2 | 65549 | 48 | 10 | 25 | 42 | 47 | 53 | 88 | 10b | 65965 | 57 | 21 | 10 | 42 | 67 | 72 | 86 |
| 3 | 65395 | 42 | 9 | 25 | 35 | 40 | 48 | 67 | 11 | 65968 | 35 | 13 | 6 | 25 | 35 | 44 | 76 |
| 4 | 65560 | 59 | 13 | 21 | 51 | 59 | 68 | 90 | 12 | 65784 | 21 | 11 | 2 | 13 | 18 | 26 | 63 |
| 5 | 65968 | 36 | 7 | 19 | 32 | 36 | 40 | 67 | 13 | 56139 | 34 | 19 | 1 | 18 | 32 | 50 | 81 |
| 6 | 57642 | 44 | 10 | 15 | 37 | 42 | 51 | 68 | 14 | 65968 | 61 | 8 | 24 | 57 | 61 | 66 | 80 |
| 7 | 65968 | 54 | 10 | 24 | 49 | 54 | 62 | 78 | 15 | 65967 | 42 | 17 | 7 | 30 | 40 | 53 | 81 |
| 8 | 65962 | 31 | 6 | 15 | 26 | 29 | 36 | 54 | 16 | 65968 | 68 | 7 | 41 | 63 | 67 | 73 | 88 |
| 9 | 29568 | 40 | 14 | 14 | 25 | 40 | 53 | 71 | 17 | 65966 | 69 | 8 | 18 | 67 | 70 | 74 | 87 |
|  |  |  |  |  |  |  |  |  | 18 | 65927 | 50 | 14 | 23 | 37 | 51 | 62 | 85 |

Lastly, a third recommendation that has been adopted into the Bin 3 and 4 simulations provides adjustments that are intended to account for how long it takes moving water to transport pesticide to the end of the end of the downstream boundary of the watershed from different starting points in the watershed. A pesticide that is deposited (via runoff or spray drift) into a headwater stream may not reach a downstream waterbody of interest until days later, while a pesticide deposited directly into the downstream waterbody is present immediately. The watersheds of Bins 3 and 4 are sufficiently large that their stream drainage networks include upstream zones that take multiple days for the pesticide and water to be transported to the end of the bin. To account for this, “time-of-travel” adjustments have been implemented, so that the watersheds associated with Bin 3 and 4 waterbodies are divided into fractions that represent the nominal number of days required to move the pesticide through the stream network of the bin. Portions of the total pesticide load (mass) introduced by a runoff or drift event are offset by time lags that reflect their nominal distance, in days, upstream from the end of the bin. Following apportionment by watershed area fraction and offsetting by the appropriate time lag, the time series from each upstream section are superimposed to generate an overall time series reflecting circumstances at the end of the bin. The time series for the runoff volume (flow) and pesticide mass are each treated in this manner. At this time hydrodynamic dispersion, or the flattening of a pesticide concentration in the direction of the flow, is not included in the simulations, although this modification of the new methodology is being considered for future BEs. Representative area fractions for Bin 3 and 4 watersheds are based on mean upstream area fractions within each HUC 2, from NHD data for reaches that have mean flowrates within 5% of the defined flowrates for Bins 3 and 4 (1 and 100 m3/s, respectively). It should be noted that, as this methodology is still under development, this refinement has not been incorporated into the BEs for the three OPs; however, EPA intends to incorporate this adjustment into the draft BEs for carbaryl and methomyl and will update this section of the draft BEs accordingly.

## 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, 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.

# 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 three areas: 1) evaluation of monitoring data, 2) use of monitoring data for quantitative purposes (*i.e.*, reasonable upper bound exposure concentration), and 3) 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 help 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 do 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 transports (*e.g.*, runoff, leaching, spray drift, and volatilization) are likely?
  + What are the weather patterns (*e.g.*, rainfall) before, during, and after pesticide application?
  + Is the sample media relevant to the habitats of concern?
  + Was the study a field-scale[[28]](#footnote-28) or general monitoring study[[29]](#footnote-29)?
  + Does the pesticide use pattern reflect the pesticide use being assessed?
* Methods
  + How often are samples collected? Is it frequent enough to estimate the desired duration of exposure?
  + Are the detection limits and limits of quantitation reported and are method recoveries reported sufficient to have confidence in the results?
  + Are 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 are samples collected? Is the sample frequency enough to estimate the desired duration of exposure?
  + What is the type of sample? Is 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 are utilized?

Additional sample media specific elements to be considered include:

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

## Use of Monitoring Data for Risk Assessment Purposes

For ESA evaluation of pesticides, general monitoring studies that provide information on pesticide concentrations on the basis of 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 to estimate pesticide concentrations (NAS 2013). Commonly, monitoring data are used qualitatively in risk assessments for characterization (*e.g.*, line of evidence in a weight-of-evidence approach). 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 considers 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 particular 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[[30]](#footnote-30),[[31]](#footnote-31)) 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.

### 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, all of 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

Several tools are being investigated by USEPA 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 not yet been vetted for use in risk assessment or presented to the Services, so this investigation is still considered preliminary.

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. 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 10,000 time series realizations are generated. For each of those time series, the 1-day peak and maximum rolling average for each of the averaging periods is calculated. A ‘bias factor’ is then calculated by comparing the 5th percentile of the estimated maximums from the simulations to the actual maximums. The “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 SEAWAVEQ model, developed by the U.S. Geological Survey (Ryberg and Vecchia, 2013[[32]](#footnote-32)). SEAWAVEQ is 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.

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: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-3)
4. USDA. 2009. Small Watershed Hydrology, Win TR-55 User Guide. USDA, Natural Resources Conservation Service. January 2009. <http://www.wcc.nrcs.usda.gov/ftpref/wntsc/H&H/WinTR55/WinTR55UserGuide.pdf> [↑](#footnote-ref-4)
5. Available: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment#aquatic> [↑](#footnote-ref-5)
6. 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-6)
7. 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-7)
8. 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-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. McKay, L., T. Bondelid, A. Rea, C. Johnston, R. Moore, and T. Dewald, 2012. NHDPlus Version 2: User Guide. Available at <ftp://ftp.horizon-systems.com/NHDPlus/NHDPlusV21/Documentation/NHDPlusV2_User_Guide.pdf> [↑](#footnote-ref-11)
12. Available: <http://www.ipmcenters.org/cropprofiles/> [↑](#footnote-ref-12)
13. Available: http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment [↑](#footnote-ref-13)
14. Atrazine Ecological Exposure Monitoring Program (AEEMP) surface water samples collected from 2011 through 2014. [↑](#footnote-ref-14)
15. Background Document in Support of the Meeting of the FIFRA Scientific Advisory Panel on the Development of the Spatial Aquatic Model for Pesticide Assessments, 2015. US Environmental Protection Agency, Washington DC. Available: https://www.federalregister.gov/articles/2015/07/21/2015-17854/fifra-scientific-advisory-panel-notice-of-public-meeting [↑](#footnote-ref-15)
16. Day, T. 1975. Longitudinal dispersion in Natural Channels. Geological Survey of Canada, Ottawa, Ontario, Canada. Water Resources Journal 11:909-918 [↑](#footnote-ref-16)
17. Kendall, C. and McDonnell, J. 1998. Isotope Tracers in Catchment Hydrology. Elsevier Science B.V., Amsterdam. 1998. [↑](#footnote-ref-17)
18. Section 5 of Background Document in Support of the Meeting of the FIFRA Scientific Advisory Panel on the Development of the Spatial Aquatic Model for Pesticide Assessments, 2015. US Environmental Protection Agency, Washington DC. Available: http://www.regulations.gov/#!documentDetail;D=EPA-HQ-OPP-2015-0424-0004 [↑](#footnote-ref-18)
19. U.S. Department of Agriculture Natural Resources Conservation Service (USDA NRCS). 2008b. 2C-5 NRCS TR-55 Methodology. Available: http://www.ctre.iastate.edu/pubs/stormwater/documents/2C-5NRCSTR-55Methodology.pdf [↑](#footnote-ref-19)
20. <http://www.regulations.gov/#!docketDetail;D=EPA-HQ-OPP-2015-0424-0015> [↑](#footnote-ref-20)
21. <http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/water/watersheds/dataset/> [↑](#footnote-ref-21)
22. Fischer, H.B. 1979. Mixing in Inland and Coastal Waters. Academic Press, London [↑](#footnote-ref-22)
23. Rutherford, J.C. 1994. River Mixing. John Wiley and Sons, Chichester [↑](#footnote-ref-23)
24. “The committee notes that in evaluating models, general monitoring data and field studies need to be distinguished. General monitoring studies (see, for example, Gilliom et al. 2007) provide information on pesticide concentration in surface water or ground water on the basis of monitoring of specific locations at specific times. The monitoring reports, however, are not associated with specific applications of pesticides under well-described conditions, such as application rate, field characteristics, water characteristics, and meteorological conditions. General monitoring data cannot be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of fate and transport models.” [↑](#footnote-ref-24)
25. <http://www2.epa.gov/ingredients-used-pesticide-products/atrazine-background-and-updates#aeemp> [↑](#footnote-ref-25)
26. “The committee notes that in evaluating models, general monitoring data and field studies need to be distinguished. General monitoring studies (see, for example, Gilliom et al. 2007) provide information on pesticide concentration in surface water or ground water on the basis of monitoring of specific locations at specific times. The monitoring reports, however, are not associated with specific applications of pesticides under well-described conditions, such as application rate, field characteristics, water characteristics, and meteorological conditions. General monitoring data cannot be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of fate and transport models.” [↑](#footnote-ref-26)
27. Wauchope, R.D. The Pesticide Content of Surface Water Draining from Agricultural Fields – A Review. Journal of Environmental Quality 7:459-472. October-December 1978 [↑](#footnote-ref-27)
28. 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-28)
29. 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-29)
30. 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-30)
31. 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-31)
32. Ryberg, K.R., and Vecchia, A.V., 2013, seawaveQ—An R package providing a model and utilities for analyzing trends in chemical concentrations in streams with a seasonal wave (seawave) and adjustment for streamflow (Q) and other ancillary variables: U.S. Geological Survey Open-File Report 2013–1255, <http://dx.doi.org/10.3133/ofr20131255> [↑](#footnote-ref-32)