Chapter 3 – Final Glyphosate Exposure Characterization

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Environmental Transport and Fate Characterization

Glyphosate has a high solubility, low octanol-water partitioning coefficient, low vapor pressure, and low Henry’s Constant (Table 3-1). These data suggest that glyphosate has a low potential for volatilization and bioaccumulation. It is assumed that the glyphosate salts dissociate rapidly to form glyphosate acid and the counter ion. Because glyphosate acid will be a zwitterion (presence of both negative (anionic) and positive (cationic) electrostatic charges) in the environment, it is expected to speciate into dissociated species of glyphosate acid as well as glyphosate-metal complexes in soil, sediment, and aquatic environments. Glyphosate can form various metal complexes (Popov et al., 2001).The formation of glyphosate-metal complexes with iron and aluminum promotes a high sorption affinity of glyphosate on the surfaces of Fe and Al oxides in soils and sediments (McBride, 1994).The main routes of dissipation are microbial degradation under aerobic conditions, and runoff. Glyphosate is expected to reach surface water primarily through spray drift; however, transport in runoff may also occur primarily via sorption of glyphosate-metal complexes to eroded sediment. The highest concentrations of glyphosate in surface water are in urban environments and in the vicinity of local use areas.

Table 3-1. Physical and Chemical Properties of Glyphosate

|  |  |
| --- | --- |
| **Property** | **Value** |
| Melting Point (°C)  | 210-212° C (tech.)215-219° C (pure) |
| Molecular Weight (g/mol) | 169.08 |
| Water Solubility@25°C mg/L | 12,000 |
| Vapor Pressure@ 25°C Torr (Pa) | 9.8x10-10(1.3 x 10-7) |
| Henrys Law Constant (calculated) atm-m3 mole-1 (pa-m3 mole-1) | 2.1x10-14(2.1 x 10-9) |
| Kow(log Kow) | <0.001(< -3) |
| Dissociation Constants | pKa1 = 0.8pKa2 = 2.35pKa3 = 5.84pKa4 = 10.48 |

In laboratory studies, glyphosate was not observed to break down by abiotic processes, such as hydrolysis, direct photolysis on soil, or photolysis in water at pH 7. The environmental fate data for glyphosate, except for a photodegradation study (MRID 44320643), did not address the impact of environmental fate processes on different species of glyphosate acid.

The major route of transformation of glyphosate identified in laboratory studies is microbial degradation. In soils incubated under aerobic conditions, the half-life of glyphosate ranges from 1.8 to 109 days and in aerobic water-sediment systems is 14 - 518 days. However, anaerobic conditions limit the metabolism of glyphosate (half-life 199 - 208 days in anaerobic water-sediment systems).

In the field, soil dissipation half-lives for glyphosate were measured to be 1.4 to 142 days. Although the variability in glyphosate dissipation rates cannot be statistically correlated to any specific test site properties, dissipation half-lives tend to be higher at test sites in the central to northern United States. Along with significant mineralization to carbon dioxide, the major metabolite of glyphosate is aminomethylphosphonic acid (AMPA).

The degradation product AMPA is a major degradation product from glyphosate. It was detected in all laboratory studies except for the abiotic hydrolysis studies. The laboratory and field dissipation data indicate that AMPA is substantially more persistent than glyphosate.

The available laboratory data indicate that both glyphosate and AMPA sorb strongly to soil. The formation of glyphosate-metal complexes promotes a high sorption affinity of glyphosate to Fe and Al oxide surfaces on soils and sediments (McBride, 1994; Popov et al., 2001). AMPA is ionic because it retains the phosphonate and amine functional groups. Because of these functional groups, AMPA will form metal complexes with Ca2+, Mg2+, Mn2+, Cu2+, and Zn2+ (Popov et al., 2001). Freundlich partitioning coefficients (Kf) for glyphosate ranged from 9.4 to 479 L/kg with exponents of 0.72 to 1, which corresponding organic carbon partitioning coefficients (Kfoc) of 1,600 to 33,000 mL/goc. Freundlich sorption coefficients for AMPA range from 10 to 509 L/kg with exponents (1/n) of 0.78 to 0.98. Because the Freundlich exponents for glyphosate and AMPA are not equal to 1, the sorption process is non-linear and, therefore, sorption coefficients are dependent on the concentration in soil solution or aquatic environments. Although this non-linearity in sorption is not captured in the exposure modeling, it is expected to reduce the exposure concentrations in aquatic exposure modeling.

Although the coefficient of variation for Kfoc is less than the coefficient of variation for Kf, indicating that pesticide binding to the organic matter fraction of the soil may explain some of the variability among the adsorption coefficients, the physicochemical properties of glyphosate (ionic) and the propensity for glyphosate and AMPA to form metal-ligand complexes on surfaces of iron and aluminum oxides would suggest the Freundlich model is the most appropriate partitioning model. This model would account for sorption on both mineral and organic constituents in soils and sediments. Based on measured Koc values, glyphosate is classified as slightly mobile to hardly mobile according to the FAO classification scheme (FAO, 2000) and is not expected to leach to groundwater or to move to surface water at high levels through dissolved runoff. However, glyphosate does have the potential to contaminate surface water from spray drift or transport of residues adsorbed to soil particles suspended in runoff. It is expected to be persistent in anaerobic sediments.

The potential for volatilization of glyphosate from soil and water is expected to be low due to the low vapor pressure and low Henry’s Law constant. Several studies have shown both glyphosate and AMPA detections in rainwater near use locations. In most cases, these detections were found during the spraying season in the vicinity of local use areas and can be attributed to spray drift rather than to volatilization or long-range transport (Baker et al., 2006; Quaghebeur et al., 2004). The highest concentrations were found in urban locations. At one site in Belgium that was 5 m from a spraying location in an urban parking lot, glyphosate was detected in rainwater for several months following a single application (Quaghebeur et al., 2004). Deposition was measured to be 205 µg a.i./m2 at one week after spraying and 0.829 µg/m2 two months after spraying. These data suggest that glyphosate can be present in the air for a long period of time (e.g., months) and this could be the result of volatilization of glyphosate from hard surfaces despite its low vapor pressure or, more likely, sorption of glyphosate to dust particles in the air or become suspended in the air.

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

Table 3-2. Environmental Fate Data for Glyphosate

|  |  |  |  |
| --- | --- | --- | --- |
| **Study** | **Value** | **Major Degradates**1**,****Comments** | **MRID #** |
| Abiotic HydrolysisHalf-life | Stable  | None | 00108192 44320642 |
| Direct Aqueous Photolysis | Stable(t1/2=216 days) | AMPA (6.6% of AR)  | 41689101 44320643 |
| Soil Photolysis Half-life | Stable(for at least 30 days) | Degradation in dark control was equal to that in irradiated samples | 44320645 |
| Aerobic Soil MetabolismHalf-life | 1.8 days (sandy loam; 25oC) 2.6 days (silt loam; 25oC)7.5 days (sandy loam; 25oC)2.04 days (sandy loam; 25oC)19.3 days (sandy loam; 20oC)27.4 days (scl loam; 20oC)7.78 days (clay loam; 20oC)109 days (silt loam; 20oC) | AMPA (24-32% of AR) CO2 (53 to >70% of AR)  | 42372501 4432064544125718PMRA1161813 Al-Rajab and Schiavon, 2010 |
| Anaerobic Aquatic MetabolismHalf-life | 208 days 203 days 199 days | AMPA (21.9-31.6% of AR)CO2 (23-35% of AR)AMPA and glyphosate were detected in sediment at 1-year posttreatment  | 41723701 4237250244125718 |
| Aerobic Aquatic MetabolismHalf-life | 14.1 days (25oC)267 days (20oC)518 days (20oC) | AMPA (25% of applied AR)CO2 (≥23% of applied AR) | 41723601; 42372503PMRA 161822 |
| **Study** | **Value** | **MRID #** |
| Batch Equilibrium | *Soil* | *KF* (L/kg) | *1/n* | *KFoc* (mL/goc) | 44320646 |
| sand | 64 | 0.75 | 22,000 |
| sandy loam | 9.4 | 0.72 | 1,600 |
| sandy loam | 90 | 0.76 | 5,000 |
| silty clay loam | 470 | 0.93 | 21,000 |
| silty clay loam | 700 | 0.94 | 33,000 |
| Silty clay loam | 62 | 0.90 | 3,172 | 00108192 |
| Silt | 90 | 0.94 | 13,050 |
| Sandy loam | 70 | 0.95 | 5,075 |
| Sandy loam | 22 | 0.78 | 5,468 |
| Sediment | 175 | 1.0 |  |
| **Study** | **Value** | **MRID #** |
| Terrestrial Field DissipationHalf-life | Glyph. AMPA1.7 d 131 d (TX)7.3 d 119 d (OH)8.3 d 958 d (GA)13 d 896 d (CA)17 d 142 d (AZ)25 d 302 d (MN)114 d 240 d (NY)142 d no data (IA) | Bare ground studies.Glyphosate and AMPA were found predominantly in the 0 to 6-inch layers | 42607501 42765001 |
| Glyph. AMPA 2.79 d 48d (CA)31 d ND (NC) | Bare ground StudiesGlyphosate and AMPA were found predominantly in the surface soil layers. | 4412571944422201 |
|  | Glyph AMPA3.9 d ND Bare ground1.4 d ND Turf  | Bare ground and turf plots in MSGlyphosate and AMPA were found predominantly in the surface soil layers. | 44320648 |
|  | Glyph AMPA19 d ND Bare ground12 d ND Turf | Bare ground and turf plots in CAGlyphosate and AMPA were found predominantly in the surface soil layers. | 4432064944320650 |
| Aquatic Field Dissipation Half-life | 7.5 d – water120 d- sediment | In a farm pond in Missouri.At 3 sites (OR, GA, MI), half-lives could not be calculated due to recharging events.  | 40881601 |
|  | Water: Dissipated rapidly immediately after treatment.Sediment: Glyphosate remained in pond sediments at ≥ 1 ppm at 1-year post treatment. | In ponds in Michigan and Oregon and a stream in GeorgiaAccumulation was higher in the pond than in the stream sediments  | 41552801 |
| Forestry Dissipation | Foliage: < 1 dayEcosystem: Glyphosate: 100 d AMPA: 118 d | 3.75 lb a.e./A, aerial application | 41552801 |
| Bioaccumulation in Fish | 0.38X in edible tissue0.63X in nonedible tissue0.52X in whole fish  |  | 41228301 |

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

Identification of Transformation Products of Concern

As mentioned, AMPA is a major degradation product from glyphosate; however, it is not considered a residue of toxicological concern based on available toxicity data. A group of surfactants, the polyethoxylated tallow amines (POEA), have been shown to be more toxic to aquatic animals than glyphosate alone. Consideration of the potential increased toxicity of formulations is considered through the selection of toxicity endpoints and is discussed further in **Chapter 2**. Estimated aquatic exposure concentrations are based on the parent glyphosate. This is based on available data as limited fate data are available to model POEA.

Measures of Aquatic Exposure

In general, maximum application rates and minimum application retreatment intervals are modeled to estimate the exposure to glyphosate based on the Joint Glyphosate Task Force Use Matrix (**APPENDIX 1-2**), unless otherwise noted. Single applications rates for ground application methods are higher than rates permitted by aerial equipment. To streamline the assessment, use scenarios were grouped based on the relevant aquatic modeling scenario.

Glyphosate-specific modeling scenarios are used for modeling each use. This includes the selection of scenarios and agronomic practices (*e.g.*, applications methods, dates). **APPENDIX 3-1** includes model use input parameters as well as the justification for selecting these parameters. The general approaches used in determining potential exposure are described below.

Aquatic Exposure Models

Aquatic exposures (surface water and benthic sediment pore water) are quantitatively estimated for representative glyphosate uses included on the Joint Glyphosate Task Force Use Matrix (**APPENDIX 1-2**) by HUC 2 Regions (Figure 3-1) and by aquatic bin (2-7). Several models are available to use to estimate pesticide concentrations in surface water. The primary model used in this assessment is the Pesticide Root Zone Model (PRZM5) and the Variable Volume Water Model (VVWM)[[1]](#footnote-2) in the Pesticides in Water Calculator (PWC). As mentioned elsewhere, the flowing aquatic bins include bin 2 low flow, bin 3 moderate flow, and bin 4 high flow. The static aquatic bins include bin 5 low volume, bin 6 moderate volume, and bin 7 high volume. Additional information on aquatic bins is available in **ATTACHMENT 1-2**.

Aquatic bin 1 represents aquatic habitats associated with terrestrial habitats (e.g., riparian zones, seasonal wetlands) and is simulated using the PRZM5/VVWM and the Plant Assessment Tool (**Section 3.5**). Aquatic bins 8 and 9 are intertidal and subtidal near shore habitats, respectively, and aquatic bin 10 is the offshore marine habitat. EFED does not currently have standard conceptual models designed to estimate EECs for these estuarine/marine systems. EFED and the Services have assigned surrogate freshwater flowing or static systems to evaluate exposure for these estuary and marine bins. Aquatic bin 5 will be used as surrogate for pesticide exposure to species in tidal pools; aquatic bins 2 and 3 will be used for exposure to species at low and high tide, and aquatic bins 4 and 7 will be used to assess exposure to marine species that occasionally inhabit offshore areas.

EFED modeled these flowing and static bins in previous Biological Evaluations (BEs) by using watershed drainage areas developed from flowing waterbody relationships and static bin runoff estimates (USEPA, 2016a; USEPA, 2016b; USEPA, 2016c). However, the results have not been as expected. In short, one would expect daily concentrations in flowing bins to be lower than those in static bins, given that water, and any pesticide, is flowing out of the system. However, in most of the modeling runs, the EECs for the flowing bins were higher, and in many cases an order of magnitude higher, than the static bins. In large part, the higher EECs were the result of modeling a large watershed using assumptions that were designed to simulate a smaller watershed. For example, it was assumed that the watershed was entirely treated on the same day. For a large watershed, it is not expected that this assumption is appropriate. For the static waterbodies, the watershed-to-waterbody ratio was not really known but was derived by determining the watershed size needed to generate runoff sufficient to fill the waterbody. This relationship relied on modeling conducted using topography, surface soil characteristics, and precipitation specific to the PWC model and scenarios considered highly vulnerable to surface water runoff. As a result, there was much uncertainty with the EECs that were being generated from the modeling in the previously completed BEs.

For glyphosate, when using PWC, EFED has relied on two standard waterbodies which have been traditionally used in EFED to estimate EECs for the various bins. The standard farm pond was used to develop EECs for the medium and large static bins (*e.g.*, bins 6 and 7) and the index reservoir for the medium and large flowing bins (*e.g.*, bins 3 and 4). For the smallest flowing and static bins (bin 2 and 5), EFED derived edge of field estimates from the PRZM daily runoff file (*e.g.*, ZTS file). Table 3-3 provides a crosswalk of the bins and how they were modeled.

Table 3-3. Aquatic Bin, Modeled Waterbody Crosswalk

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Aquatic Bin** | **Description** | **Width (m)** | **Length (m)** | **Depth (m)** | **Flow (m3/s)** | **Waterbody Used for Modeling** |
| 1 | Wetland | 64 | 157 | 0.15 | Variable1 | Custom |
| 2 | Low-flowing waterbody | 2 | Field2 | 0.1 | 0.001 | Edge-of-field |
| 3 | Medium-flowing waterbody | 8 | Field2 | 1 | 1 | Index reservoir |
| 4 | High-flowing waterbody | 40 | Field2 | 2 | 100 | Index reservoir |
| 5 | Low-volume, static waterbody | 1 | 1 | 0.1 | N/A | Edge-of-field |
| 6 | Medium-volume, static waterbody | 10 | 10 | 1 | N/A | Farm pond |
| 7 | High-volume, static waterbody | 100 | 100 | 2 | N/A | Farm pond |

1 The depth and flowrate in this waterbody is variable, depending on rainfall.

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

While the standard farm pond is bigger than bin 6, the EECs estimated for bin 6 in previous BEs were close to those generated for bin 7, and so an economy of modeling was deemed appropriate.

While the index reservoir has a much lower effluent flowrate than bins 3 and 4, it has been used as a vetted flow-through waterbody for EFED for years, with an accepted watershed-to-waterbody ratio developed for an actual vulnerable watershed (Shipman Reservoir, Shipman, IL) and has been reviewed by a previous Federal Insecticide Fungicide Rodenticide Act (FIFRA) Scientific Advisory Panel (SAP) (Jones *et al*, 1998). EFED expects the EECs that are generated using the index reservoir to be a conservative surrogate for those observed in bins 3 and 4. The watershed area associated with the index reservoir is roughly an order of magnitude smaller than the average area for a HUC 12 (the smallest areal delineation for an aquatic species range), but within the range of minimum and maximum values (9.54x107 m2, 2.08x103 – 9.24x109 m2).

Lastly, bins 2 and 5 are very small waterbodies and the EECs in them would be reflective of concentrations in a headwater stream or a standing puddle that received runoff at the edge of a treated field. As such, edge-of-field concentrations were estimated and used as a surrogate for EECs in these waterbodies.

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

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

HUC and Use Site Crosswalk

The National Agricultural Statistics Census of Agriculture 2012 (NASS) data along with the Cropland Data Layer (CDL) were used to determine which crops would be modeled within each represented HUC 2. Additionally, specific geographic limitations on how a product may be applied to particular crops were considered when determining what rates would be simulated for different HUC 2 regions. For example, spray drift buffers are required in some states (i.e., California and Arkansas) for some applications (e.g., rice, sugarcane, and forestry).

If the NASS data indicated any acreage of a crop was grown in a specific HUC 2, it was assumed that the crop was grown in that HUC 2, and aquatic EECs were generated for these HUC 2 regions for that crop. If there were no reported NASS cropped acres grown within a particular HUC 2, aquatic EECs for that HUC 2 region and use pattern were not determined. A crop use layer-HUC 2 region matrix for glyphosate is provided in **APPENDIX 3-1**. Limited NASS data are available for Alaska, Hawaii, and Puerto Rico, and some assumptions on which crops would be simulated in those HUC 2 regions were made.

Scenario Selection

A PWC scenario was developed for each landcover class and HUC 2 where crops in that landcover were grown based on the NASS 2012 census data. **APPENDIX 3-1** provides a crosswalk between the use site and the landcover used to represent the use site as well as which HUC 2 regions were evaluated for each use pattern. An explanation of how the PWC scenario matrix was developed is provided in **ATTACHMENT 3-1**.

Application Practices

Application Method

During application of pesticides, methods of application as well as product formulation used by an applicator can impact the magnitude of off-site transport of the active ingredient. Label directions (such as spray drift buffers and droplet size restrictions, application equipment, and agronomic practices such as soil incorporation) as well as product formulation are considered as part of the development of the use scenario modeled.

There are several different types of glyphosate applications included in the Joint Glyphosate Task Force Use Matrix (**APPENDIX 1-2**) including those that occur in both agricultural and non-agricultural settings. Applications occur from aircraft, boats, pick-ups, and tractors and application equipment include: boom sprayer, hand held hydraulic sprayer, hand held sprayer, high volume ground sprayer, hooded sprayer, hose-end sprayer, low volume ground sprayer, low volume sprayer, motor driven sprayer, product container, ready-to-use spray container, shielded applicator, sprayer, tank-type sprayer, wick applicator, and wiper applicator). Glyphosate is formulated as water-dispersible granules (WG) (80% active ingredient), emulsifiable concentrate (EC) (13.4% - 36.5% active ingredient), water-dispersible liquids (L) (5% - 14.6% active ingredient), ready to use (RTU) (0.81% active ingredient), and soluble concentrate/solid (SC/S) (95.2% - 96.7% active ingredient).

Glyphosate applications may occur at different times throughout the year including multiple applications to the same crop. When multiple types of applications are allowed on a crop within one calendar year, such as pre-plant along with post-harvest applications, all applications are simulated considering the appropriate application timing (*e.g.*, dormant, foliar, and post-harvest applications to a crop) and label directions.

For perennial crops such as orchards, stump applications are only made at the end of the crop cycle which can range from 15 to 50 years[[2]](#footnote-3). As such, these application scenarios were not modeled as the current modeling (yearly repeated application capture the range of potential concentration from sporadic applications).

Spray Drift

The default spray drift inputs were assumed for all glyphosate applications except for shielded or wiper applications and in states (i.e., California and Arkansas) where buffers are required for specific uses (e.g., rice, sugarcane, and forestry). For example, for shielded or wiper applications drift was assumed to be 0 while the default application efficacy for ground application (i.e., 0.99). The default spray drift fractions used for all foliar spray in modeling are shown in **Table 3-4**. No spray drift is simulated for the shielded or wiper applications.

Table 3-4. Estimated Spray Drift Fractions for Different Aquatic Bins and Application Methods

|  |  |
| --- | --- |
| **Bin** | **Spray Drift Fraction****(unitless)** |
| **Aquatic Bin** | **PWC Bin Number** | **Generic Habitat** | **Depth (m)1** | **Width (m)1** | **Default** |
| **Aerial** | **Ground** |
| 4 | 4 | Reservoir | 2.74 | 82 | 0.135 | 0.066 |
| 7 | 7 | Pond | 2 | 64 | 0.125 | 0.062 |
| 1 | 10 | Wetland | 0.15 | 64 | 0.125 | 0.062 |
| 1parameters correspond to the input values used in PWC modeling.EOF concentrations from bin 4 are used as a surrogate for aquatic bins 2 and 5.Aquatic bin 4 is used as a surrogate for aquatic bin 3.  |

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

Application Timing

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

Glyphosate may be applied during different seasons, and the directions for use indicate the timing of application, such as at plant, dormant season, foliar (*i.e*., when foliage is on the plant), *etc.* For most glyphosate uses, PWC model inputs for the application dates were chosen based on these timings, the crop emergence and harvest timings specified in the PWC scenario, and precipitation data for the associated meteorological station. Application dates were selected to represent conservative and reasonable estimates. If applicable, dormant seasons were assumed to occur between November and February, the predominant period throughout the country when crops are dormant. Foliar applications were assumed to occur when the crop was on the field in the PWC scenario for glyphosate resistant or GMO crops; however, non-GMO applications were assumed to occur prior to planting or after harvest. When choosing an application date within a time window (*i.e*., crop emergence or foliar application), the first or 15th of the month with the highest amount of precipitation (for the meteorological station for the PWC scenario) for that time window was chosen. Pesticide loading to surface water is directly affected by precipitation events. Once the first day of application was selected, minimum retreatment intervals were assumed to determine when subsequent applications would occur. If multiple types of applications were allowed on one crop within one year, such as pre-plant or soil incorporation along with a foliar application(s), the retreatment interval was selected to reflect the specified timings. Pre-harvest intervals (the minimum time between an application and harvest) were also considered. Applications would not occur closer to harvest than allowed by the pre-harvest interval. For details on application date selection for use of glyphosate, see **APPENDIX 1-3** and **APPENDIX 3-1**.

Special Agricultural Considerations

Multiple Crop-cycles Per Year

Some labels permit applications on crops that may be planted in rotation (*e.g.*, wheat), or that may be grown in multiple crop seasons per year. In addition, fallow applications are permitted. However, since the total use of glyphosate is restricted on an annual basis these agricultural practices should not result in more than 6 or 8 lb of glyphosate per acre per year for agricultural uses.

PFAM

Glyphosate is used on rice and cranberry and it is common practice to flood rice fields and cranberry bogs. Water from flooding a rice patty or cranberry bog is generally released to adjacent waterbody (wetlands, cannels, rivers, streams, lakes, or bays). The Pesticides in Flooded Applications Model (PFAM, version 1.09) was used to simulate rice and cranberry agricultural practices.

PFAM was developed specifically for regulatory applications to estimate exposure for pesticides used in flooded agriculture such as rice paddies and cranberry bogs. The model considers the environmental fate properties of pesticides and allows for the specifications of common management practices that are associated with flooded agriculture such as scheduled water releases and refills. It estimates both acute and chronic concentrations over different durations, allows for defining different receiving water bodies.

PFAM will be used to estimate the concentration of glyphosate in the flood water released from a rice patty or cranberry bog. The reported concentrations represent water introduced to the field and not mixed with any additional water (*i.e.*, receiving water body). The concentration of glyphosate is expected to be more than what would be expected in adjacent water bodies due to additional degradation and dilution unless the water is released into an empty canal, ditch, etc. The difference in the concentration of glyphosate in the flood water to that in an adjacent waterbody depends on 1) the length of time glyphosate is in the flooded patty or bog, 2) the distance the water travels between the patty or bog and the adjacent waterbody, 3) the amount of dilution and 4) whether the flood water is mixed with additional water that also contains glyphosate. PFAM can simulate application of a pesticide to a dry field and degradation in soil before water is introduced to the bog.

Plant Assessment Tool (PAT)

The Plant Assessment Tool (PAT) is a mechanistic model that incorporates fate (e.g., degradation) and transport (e.g., runoff) data that are typically available for conventional pesticides, to estimate pesticide concentrations in terrestrial, wetland, and aquatic plant habitats. For terrestrial plants, runoff and erosion are modeled using PRZM and spray drift is modeled using AgDRIFT deposition values. The model uses a mixing cell approach to represent water within the active root zone area of soil, and accounts for flow through the terrestrial plant exposure zone (T-PEZ) caused by both treated field runoff and direct precipitation onto the T-PEZ. Pesticide losses from the T-PEZ occur from transport (i.e., washout and infiltration below the active root zone) and degradation. Wetlands are modeled using PRZM/VVWM and are then processed in PAT to estimate aquatic (mass per volume of water) and terrestrial (mass per area) concentrations. Aquatic plants exposure is modeled using the PRZM/VVWM models and the standard farm pond.

Direct Water Applications

The glyphosate labels with direct aquatic uses do not specify a target concentration as the aquatic label application rates are expressed as lb ae/A. Direct applications to water were considered for glyphosate using an equilibrium partitioning equation considering a single-compartment first-order transformation rate. The equation used is provided in Figure 3-2. Aerobic aquatic metabolism and sorption were assumed to control the dissipation rate of glyphosate.

Cw = (mai′)e-kt / (dw + dsed(θsed + ρbKd)

and,

Kd= 0.01 Koc

where: Cw = water concentration (µg/L)

mai′ = mass applied per unit area kg/ha

k = transformation rate constant (d-1)

t= time elapsed since application (d)

Kd = water-sediment partitioning coefficient (L/kg)

Koc = organic carbon partitioning coefficient (L/kg)

Figure 3-2. Equation Used to Estimate Concentration of Glyphosate Following Direct Application to Water

Non-Agricultural Uses and Considerations

As described in the Joint Glyphosate Task Force Use Matrix (**APPENDIX 1-2**) there is a non-agricultural use site, including residential lawns, forestry commercial and recreational complexes.

Aquatic Modeling Input Parameters

The following sections discuss methods used for aquatic modeling.  Summaries of the environmental fate model input parameters used in the PWC for the modeling of glyphosate is presented in Table 3-4. Input parameters are selected in accordance with the following EPA guidance documents:

* Guidance for *Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides*, Version 2.1[[3]](#footnote-4) (USEPA, 2009),
* *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[4]](#footnote-5)* (NAFTA, 2012; USEPA, 2012c), and
* *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[5]](#footnote-6) (USEPA, 2013)

Table 3-4. Input Values Used for Tier II Surface Water Modeling with PWC or PFAM

| **Parameter (units)** | **Value (s)** | **Source** | **Comments** |
| --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (Kf (L/kg)) | 175 | MRIDs 44320646, 00108192 | Mean Kf (64, 9.4, 90, 470, 700 (MRID 44320646), 62,90, 70, 22, and 175 (MRID 00108192)a |
| Water Column Metabolism Half-life or Aerobic Aquatic Metabolism Half-life (days) 25 °C | 381 | MRID 41723601PMRA 1161822 | Upper 90th percentile confidence bound of the mean half-life=189.7+(1.886\*175.8)/SQR(3)Average=189.7SD=175.8T n-1,90 =1.886n=3 (14, 188a and 366a) |
| Benthic Metabolism Half-life or Anaerobic Aquatic Metabolism Half-life (days) 25 °C | 208 | MRID 41723701MRID 42372502 | Upper 90th percentileconfidence bound of the meanhalf-life=203.33+(1.886\*4.509)/SQR(3)Average=203.33SD=4.509T n-1,90 =1.886n=3 |
| Aqueous Photolysis Half-life @ pH 7 (days) and Reference Latitude 40o N latitude, 25oC | Stable (0) | MRID 41689101, 44320643 | Represents photo-degradation rate at pH 7 |
| Hydrolysis Half-life (days) | Stable | MRIDs 00108192, 44320642 |  |
| Soil Half-life or Aerobic Soil Metabolism Half-life (days) 25 °C | 29 | MRIDs 44320645, 44125718, 42372501 PMRA 1161813Al-Rajab and Schiavon, 2010 | Upper 90th percentile confidence bound of the mean half-life=16.19+(1.415\*25.37)/SQR(8)Average=16.19SD=25.37T n-1,90 =1.415n=8 (1.8, 2.0, 2.6, 5.5b, 7.5, 13.6b, 19.4b, 77.1b) |
| MWT or Molecular Weight (g/mol) | 169.08 | Calculated |  |
| Vapor Pressure (Torr) at 25oC | 9.75E-10 |  |  |
| Solubility in Water @ 25oC, pH not reported (mg/L) | 12,000 | Product Chemistry |  |
| Foliar Half-life (days) | 0 | default |  |
| Application Efficiency | 1 (ground - shield)0.99 (ground)0.95 (air) | default |  |
| Drift | Default | AgDRIFT | See section 3.4.2 |
| Heat of Henry | 49,884 | EPIweb |  |
| 1. Although the coefficient of variation for Kfoc is less than the coefficient of variation for Kf, indicating that pesticide binding to the organic matter fraction of the soil may explain some of the variability among the adsorption coefficients, the physicochemical properties of glyphosate (ionic) and the propensity for glyphosate and AMPA to form metal-ligand complexes on surfaces of iron and aluminum oxides would suggest the Freundlich model is the most appropriate partitioning model. This model would account for sorption on both mineral and organic constituents in soils and sediments.
2. Half-lives corrected from 20oC to 25oC using Q10 temperature correction equation.
 |

Aquatic Modeling Results

The estimated environmental concentrations (EECs) derived from the PWC modeling based on maximum labeled rates included in the Joint Glyphosate Task Force Use Matrix, by HUC 2, are summarized for the various aquatic bins in Table 3-5and Table 3-6, for water column and pore water, respectively. EECs for direct applications and rice and cranberry applications are summarized in Table 3-7andTable 3-8, respectively. The complete set of modeling inputs and results are available in **APPENDIX 3-1**. PWC runs and post-processed results are provided in **APPENDIX 3-2**. The range of EECs reflects the various application rates and timings for the various crops and scenarios modeled in the HUC 2 regions.

Table 3-5. The Range of PWC Daily Average Water Column EECs for Glyphosate

| **HUC 2** | **Range of 1-in-15 year Daily Average EECs (µg/L)** |
| --- | --- |
| **Bin 1** | **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| HUC 1 | 54.5 - 1280 | 11 – 19,796 | 17.17 – 1,130 | 17.17 – 1,130 | 11 – 19,796 | 17.61 – 1,246 | 17.61 – 1,246 |
| HUC 2 | 54.3 - 1370 | 9 – 16,110 | 6.309 – 1,045 | 6.309 – 1,045 | 9 – 16,110 | 8.141 – 1,245 | 8.141 – 1,245 |
| HUC 3 | 53.5 - 1200 | 9 – 16,135 | 11.41 – 1,637 | 11.41 – 1,637 | 9 – 16,135 | 12.45 – 1,718 | 12.45 – 1,718 |
| HUC 4 | 57.2 - 1370 | 10 – 18,972 | 7.877 – 1,824 | 7.877 – 1,824 | 10 – 18,972 | 8.7 – 1,941 | 8.7 – 1,941 |
| HUC 5 | 54.1 - 1320 | 9 – 17,795 | 16.43 – 1,693 | 16.43 – 1,693 | 9 – 17,795 | 15.81 – 2,056 | 15.81 – 2,056 |
| HUC 6 | 53.4 - 1100 | 9 – 20,226 | 10.23 – 1,202 | 10.23 – 1,202 | 9 – 20,226 | 9.772 – 1,355 | 9.772 – 1,355 |
| HUC 7 | 54.4 - 1360 | 9 – 19,749 | 18.02 – 1,537 | 18.02 – 1,537 | 9 – 19,749 | 17.75 – 1,522 | 17.75 – 1,522 |
| HUC 8 | 52.5 - 1330 | 10 – 19,832 | 10.02 – 1,877 | 10.02 – 1,877 | 10 – 19,832 | 12.79 – 1,584 | 12.79 – 1,584 |
| HUC 9 | 54.6 - 1380 | 14 – 19,084 | 12.71 – 1,248 | 12.71 – 1,248 | 14 – 19,084 | 14.58 – 1,573 | 14.58 – 1,573 |
| HUC 10a | 54.3 - 1380 | 11 – 16,879 | 16.95 – 1,542 | 16.95 – 1,542 | 11 – 16,879 | 13.89 – 1,577 | 13.89 – 1,577 |
| HUC 10b | 54.5 - 1370 | 11 – 17,114 | 13.36 – 1,082 | 13.36 – 1,082 | 11 – 17,114 | 12.79 – 1,023 | 12.79 – 1,023 |
| HUC 11a | 53.4 - 1360 | 11 – 15,196 | 11.61 – 1,897 | 11.61 – 1,897 | 11 – 15,196 | 14.03 – 1,755 | 14.03 – 1,755 |
| HUC 11b | 53.9 - 1370 | 10 – 14,492 | 10.09 – 1,545 | 10.09 – 1,545 | 10 – 14,492 | 12.36 – 1,761 | 12.36 – 1,761 |
| HUC 12a | 53.2 - 1330 | 15 – 20,311 | 10.53 – 1,995 | 10.53 – 1,995 | 15 – 20,311 | 12.25 – 1,513 | 12.25 – 1,513 |
| HUC 12b | 53.4 - 1380 | 15 – 16,588 | 10.98 – 1,527 | 10.98 – 1,527 | 15 – 16,588 | 11.7 – 1,328 | 11.7 – 1,328 |
| HUC 13 | 11.8 - 1380 | 15 – 19,351 | 9.477 – 1,788 | 9.477 – 1,788 | 15 – 19,351 | 10.13 – 1,071 | 10.13 – 1,071 |
| HUC 14 | 54.3 - 1380 | 14 – 18,652 | 12.83 - 900.1 | 12.83 - 900.1 | 14 – 18,652 | 12.93 - 989.5 | 12.93 - 989.5 |
| HUC 15a | 54.8 - 1380 | 14 – 18,092 | 11.99 – 1,163 | 11.99 - 1163 | 14 – 18,092 | 15.09 – 1,246 | 15.09 – 1,246 |
| HUC 15b | 4.23 - 1370 | 13 – 18,922 | 8.325 - 542.5 | 8.325 - 542.5 | 13 – 18,922 | 6.111 - 358.4 | 6.111 - 358.4 |
| HUC 16a | 44.9 - 1380 | 13 – 17,834 | 10.66 – 1,220 | 10.66 - 1220 | 13 – 17,834 | 11.91 – 1,131 | 11.91 – 1,131 |
| HUC 16b | 38.9 - 1380 | 12 – 17,018 | 11.31 - 623.2 | 11.31 - 623.2 | 12 – 17,018 | 5.101 - 612.6 | 5.101 - 612.6 |
| HUC 17a | 54.5 - 1230 | 14 – 17,846 | 15.88 – 1,223 | 15.88 - 1223 | 14 – 17,846 | 17.24 – 1,452 | 17.24 – 1,452 |
| HUC 17b | 33.8 - 1380 | 14 – 15,892 | 12.68 - 401.7 | 12.68 - 401.7 | 14 – 15,892 | 5.177 - 468.5 | 5.177 - 468.5 |
| HUC 18a | 6.69 - 1360 | 8 – 16,856 | 9.538 – 1,093 | 9.538 - 1093 | 8 – 16,856 | 10.85 – 1,083 | 10.85 – 1,083 |
| HUC 18b | 4.71 - 1370 | 7 – 18,245 | 9.385 – 1,296 | 9.385 - 1296 | 7 – 18,245 | 10.36 - 940.1 | 10.36 - 940.1 |
| HUC 19a | 46.6 - 1390 | 17 – 18,330 | 17.5 - 788.3 | 17.5 - 788.3 | 17 – 18,330 | 11.53 - 868.6 | 11.53 - 868.6 |
| HUC 19b | 38.1 - 1240 | 17 – 19,208 | 21.6 - 527 | 21.6 - 527 | 17 – 19,208 | 15.46 - 834.4 | 15.46 - 834.4 |
| HUC 20a | 52.9 - 1170 | 17 – 20,274 | 22.59 – 2,282 | 22.59 – 2,282 | 17 – 20,274 | 31.87 – 2,027 | 31.87 – 2,027 |
| HUC 20b | 52.9 - 1370 | 12 – 20,157 | 10.63 – 1,065 | 10.63 – 1,065 | 12 – 20,157 | 11.47 - 919.6 | 11.47 - 919.6 |
| HUC 21 | 53 - 1190 | 17 – 17,765 | 14.33 – 1,203 | 14.33 – 1,203 | 17 – 17,765 | 15.54 - 810.2 | 15.54 - 810.2 |

Table 3-6. The Range of PWC Pore Water EECs for Glyphosate

| **HUC 2** | **Range of 1-in-15 year Daily Average EECs (µg/L)** |
| --- | --- |
| **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| HUC 1 | 54.5 – 1,280 | 8.36 - 611.2 | 8.36 - 611.2 | 30 – 19,796 | 10.02 - 989.9 | 10.02 - 989.9 |
| HUC 2 | 54.3 – 1,370 | 2.193 - 341.9 | 2.193 - 341.9 | 12 – 16,110 | 2.687 - 972.7 | 2.687 - 972.7 |
| HUC 3 | 53.5 – 1,200 | 5.298 - 500.3 | 5.298 - 500.3 | 18 – 16,135 | 6.668 - 1351 | 6.668 - 1351 |
| HUC 4 | 57.2 – 1,370 | 2.597 - 709 | 2.597 - 709 | 13 – 18,972 | 3.251 - 1640 | 3.251 - 1640 |
| HUC 5 | 54.1 – 1,320 | 7.922 - 472.9 | 7.922 - 472.9 | 30 – 17,795 | 8.487 - 1665 | 8.487 - 1665 |
| HUC 6 | 53.4 – 1,100 | 3.66 - 373 | 3.66 - 373 | 18 – 20,226 | 3.463 - 1061 | 3.463 - 1061 |
| HUC 7 | 54.4 – 1,360 | 9.479 - 397.9 | 9.479 - 397.9 | 30 – 19,749 | 10.01 - 1214 | 10.01 - 1214 |
| HUC 8 | 52.5 – 1,330 | 4.592 - 396.9 | 4.592 - 396.9 | 15 – 19,832 | 3.854 - 1032 | 3.854 - 1032 |
| HUC 9 | 54.6 – 1,380 | 7.848 - 540.1 | 7.848 - 540.1 | 17 – 19,084 | 8.93 - 1206 | 8.93 - 1206 |
| HUC 10a | 54.3 – 1,380 | 8.355 - 435.2 | 8.355 - 435.2 | 30 – 16,879 | 6.534 - 1159 | 6.534 - 1159 |
| HUC 10b | 54.5 – 1,370 | 7.767 - 457.8 | 7.767 - 457.8 | 30 – 17,114 | 5.091 - 767.3 | 5.091 - 767.3 |
| HUC 11a | 53.4 – 1,360 | 4.578 - 363 | 4.578 - 363 | 15 – 15,196 | 6.575 - 1209 | 6.575 - 1209 |
| HUC 11b | 53.9 – 1,370 | 5.793 - 502.8 | 5.793 - 502.8 | 15 – 14,492 | 5.67 - 1251 | 5.67 - 1251 |
| HUC 12a | 53.2 – 1,330 | 4.342 - 546.1 | 4.342 - 546.1 | 15 – 20,311 | 5.541 - 987 | 5.541 - 987 |
| HUC 12b | 53.4 – 1,380 | 4.936 - 676.2 | 4.936 - 676.2 | 15 – 16,588 | 5.869 - 1166 | 5.869 - 1166 |
| HUC 13 | 11.8 – 1,380 | 5.015 - 690.3 | 5.015 - 690.3 | 15 – 19,351 | 5.064 - 824.9 | 5.064 - 824.9 |
| HUC 14 | 54.3 – 1,380 | 8.67 - 453.7 | 8.67 - 453.7 | 15 – 18,652 | 4.035 - 809.5 | 4.035 - 809.5 |
| HUC 15a | 54.8 – 1,380 | 7.509 - 410.4 | 7.509 - 410.4 | 15 – 18,092 | 9.638 - 959 | 9.638 - 959 |
| HUC 15b | 4.23 – 1,370 | 2.657 - 257.1 | 2.657 - 257.1 | 14 – 18,922 | 0.8984 - 210.7 | 0.8984 - 210.7 |
| HUC 16a | 44.9 – 1,380 | 6.228 - 498.4 | 6.228 - 498.4 | 14 – 17,834 | 6.639 - 840.6 | 6.639 - 840.6 |
| HUC 16b | 38.9 – 1,380 | 5.34 - 493.1 | 5.34 - 493.1 | 14 – 17,018 | 1.784 - 520.6 | 1.784 - 520.6 |
| HUC 17a | 54.5 – 1,230 | 6.001 - 594 | 6.001 - 594 | 30 – 17,846 | 8.176 - 1212 | 8.176 - 1212 |
| HUC 17b | 33.8 – 1,380 | 4.61 - 258.9 | 4.61 - 258.9 | 28 – 15,892 | 1.598 - 408.3 | 1.598 - 408.3 |
| HUC 18a | 6.69 – 1,360 | 5.548 - 692.1 | 5.548 - 692.1 | 24 – 16,856 | 5.991 - 833.6 | 5.991 - 833.6 |
| HUC 18b | 4.71 – 1,370 | 5.466 - 590.8 | 5.466 - 590.8 | 24 – 18,245 | 5.56 - 697.9 | 5.56 - 697.9 |
| HUC 19a | 46.6 – 1,390 | 12.28 - 351.7 | 12.28 - 351.7 | 30 – 18,330 | 5.664 - 753.2 | 5.664 - 753.2 |
| HUC 19 b | 38.1 – 1,240 | 8.186 - 237.2 | 8.186 - 237.2 | 30 – 19,208 | 9.654 - 705.1 | 9.654 - 705.1 |
| HUC 20a | 52.9 – 1,170 | 8.614 - 366.8 | 8.614 - 366.8 | 30 – 20,274 | 10.44 - 1376 | 10.44 - 1376 |
| HUC 20b | 52.9 – 1,370 | 4.688 - 450.8 | 4.688 - 450.8 | 30 – 20,157 | 4.548 - 643.3 | 4.548 - 643.3 |
| HUC 21 | 53 – 1,190 | 5.2 - 326.4 | 5.2 - 326.4 | 30 – 17,765 | 5.194 - 505.3 | 5.194 - 505.3 |

Table 3-7. Direct Water Application EECs

|  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **Aquatic Bin** | **Description** | **Sediment depth (m)** | **Width (m)** | **Length (m)** | **Depth (m)** | **Flow (m3/s)** | **1-day** | **4-day** | **1-day** | **4-day** |
| **3.75 lb/A** | **8.0 lb/A** |
| 1 | Wetland | 0.15 | 64 | 157 | 0.15 | Variable1 | 12 | 12 | 26 | 26 |
| 2 | Low-flowing waterbody | 0.01 | 2 | Field2 | 0.1 | 0.001 | 177 | 175 | 377 | 374 |
| 3 | Medium-flowing waterbody | 0.05 | 8 | Field2 | 1 | 1 | 34 | 34 | 72 | 72 |
| 4 | High-flowing waterbody | 0.05 | 40 | Field2 | 2 | 100 | 31 | 31 | 67 | 66 |
| 5 | Low-volume, static waterbody | 0.01 | 1 | 1 | 0.1 | N/A | 177 | 175 | 377 | 374 |
| 6 | Medium-volume, static waterbody | 0.05 | 10 | 10 | 1 | N/A | 34 | 34 | 72 | 72 |
| 7 | High-volume, static waterbody | 0.05 | 100 | 100 | 2 | N/A | 31 | 31 | 67 | 66 |

1 The depth and flowrate in this waterbody is variable, depending on rainfall.

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

3 Sediment organic fraction (0.01 m); sediment bulk density (1300 kg/m3); sediment porosity (0.509)

Table 3-8. PFAM EECs

|  |  |  |  |
| --- | --- | --- | --- |
| **Scenario****Application Date** **(Rate lb a.i./A)** | **HUC 2** | **State** | **Daily Average EECs (µg/L)** |
| **Rice** |
| ECO AR noWinter4/19 (3.75), 4/26 (2.25)  | 11, 08, 07 | Arkansas | 47.1 |
| ECO CA Winter4/18 (3.75), 4/25 (2.25)  | 16, 17, 18 | California | 54.5 |
| ECO LA Winter3/27 (3.75), 4/3 (2.25) | 08 | Louisiana | 48.3 |
| ECO MO noWinter4/21 (3.75), 4/28 (2.25) | 07, 05, 06, 08, 11 | Missouri | 50.6 |
| ECO MS noWinter (winter)5/2 (3.75), 5/9 (2.25) | 03, 08 | Mississippi | 51.0 |
| ECO TX Winter4/2 (3.75), 4/9 (2.25) | 12 | Texas | 52.6 |
| **Cranberry** |
| MA\_Cranberry\_Winter Flood\_SCH3/15 (3.75), 3/22 (3.75), 3/29 (0.5) | 02 | Massachusetts | 2.3 |
| OR\_Cranberry\_No Flood\_SCH5/15 (3.75), 5/22 (3.75), 5/29 (0.5) | 17 | Oregon | 8.4 |
| OR\_Cranberry\_Winter Flood\_SCH5/15 (3.75), 5/22 (3.75), 5/29 (0.5) | 17 | Oregon | 7.9 |
| WI\_Cranberry\_Winter Flood\_SCH5/15 (3.75), 5/22 (3.75), 5/29 (0.5) | 07 | Wisconsin | 4.0 |

Available Monitoring Data

Examination of the EPA 303(d) list of impaired waters[[6]](#footnote-7) on August 31, 2020, indicates no impairments caused by glyphosate.

Water monitoring data were obtained from the National Water Quality Monitoring Council’s Water Quality Data Portal (USEPA and USGS, 2013), which is supplied by the USGS NWIS and EPA STORET databases of monitoring data collected across the United States by numerous federal, state, tribal and local agencies. Prominent contributors to this database include the USGS National Water-Quality Assessment Program (NAWQA) and various state and local agencies.

In general, the surface water monitoring data include sampling sites that represent a range of aquatic environments including small and large water bodies, rivers, reservoirs, and urban and agricultural locations, but are limited for some areas of the United States where glyphosate use occurs. Also, the sampling sites, as well as the number of samples, vary by year. The vulnerability of the sampling site to glyphosate contamination varies substantially due to use, soil characteristics, weather and agronomic practices. Often, monitoring programs in the Water Quality Portal are not specifically designed to target glyphosate use; as such, peak concentrations of glyphosate likely went undetected in these programs. Overall, the extent to which historical values represent current agronomic or labeled use instructions is uncertain.

While there are many individual samples collected and analyzed for glyphosate across the United States, it would not be appropriate to combine these data sources to generate exposure estimates or to use these datasets to represent exposure on a national or even regional basis. While these data demonstrate potential exposure, using the measured concentrations as an upper bound exposure estimate would not be a reasonable approach for the reasons given above, including limited sample frequency, limited use information, and sampling site variability, on a national or even a regional basis. Therefore, model estimated concentrations should be considered a suitable upper bound concentration for glyphosate.

Water Quality Portal

Comprehensive surface water and groundwater glyphosate data were obtained in August 2020 in a download of data from the Water Quality Data Portal (<http://www.waterqualitydata.us/>). Table 3-9 provides a summary of the results by HUC 2 region, with sampling occurring from 1975 to 2020 at over 17,500 sites in every HUC 2 region.

Table 3-9. Water Quality Portal Monitoring Data Summarized by 2-digit HUC for Glyphosate.

| **HUC-21** | **Years** | **Number of Sites** | **Number of Samples** | **Number of Samples Labeled Non-Detections** | **Measured Detection Range (µg/L)** |
| --- | --- | --- | --- | --- | --- |
| 1 | 2002 - 2020 | 29 | 314 | 241 | 0.02 - 0.91 |
| 2 | 2002 - 2020 | 49 | 563 | 417 | 0.02 - 0.83 |
| 3 | 1996 - 2020 | 769 | 3133 | 1663 | 0.02 - 200 |
| 4 | 2000 - 2019 | 63 | 678 | 409 | 0 - 7.4 |
| 5 | 1999 - 2020 | 88 | 1584 | 1017 | 0 - 4.9 |
| 6 | 2013 - 2019 | 17 | 114 | 94 | 0.02 - 0.46 |
| 7 | 1999 - 2020 | 328 | 2904 | 1780 | 0 - 42.8 |
| 8 | 2000 - 2020 | 32 | 1003 | 72 | 0.02 - 200 |
| 9 | 2006 - 2020 | 55 | 183 | 87 | 0 - 10.3 |
| 10 | 2000 - 2020 | 149 | 1686 | 449 | 0 - 50 |
| 11 | 2002 - 2020 | 10 | 229 | 13 | 0.02 - 7.8 |
| 12 | 1999 - 2020 | 11 | 279 | 27 | 0.02 - 53.5 |
| 13 | 1995 - 2020 | 23 | 223 | 96 | 0.02 - 2 |
| 14 | 1998 - 2020 | 25 | 168 | 81 | 0.02 - 0.7 |
| 15 | 2002 - 2020 | 24 | 109 | 60 | 0.02 - 2.5 |
| 16 | 1998 - 2019 | 18 | 271 | 34 | 0.02 - 4.2 |
| 17 | 1996 - 2020 | 409 | 2523 | 1299 | 0 - 45.8 |
| 18 | 1994 - 2020 | 517 | 1904 | 1237 | 0 - 590 |
| 19 | 2014 - 2015 | 1 | 6 | 6 | NA |
| 20 | 1990 - 2014 | 10 | 10 | 8 | 0.03 - 0.03 |
| 21 | 2013 | 2 | 2 | 0 | 0.1 - 0.21 |
| 22 | 2002 - 2020 | 29 | 314 | 241 | 0.02 - 0.91 |

1 Note that historical monitoring data does not reflect updates to labels or commitment letters (for example, data in HUCs 19-22)

## Aquatic Exposure Summary

Model-derived EECs represent an upper bound on potential exposure as a result of the use of glyphosate. Comparing the concentrations in the medium and high flowing and static bins (3, 4, 6, and 7) to the highest measured concentrations, the modeled values are much higher than the measured concentrations. The medium and high flowing and static bins were used for comparison purposes as these would seem to represent typical waterbodies considered for ambient water monitoring, as they typically have flow and water present all year. As recommended by the NRC in the 2013 NAS report, general monitoring data are not recommended to be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of EPA’s fate and transport models. However, EPA believes monitoring data can be used as part of the weight-of-evidence evaluation to evaluate potential exposure.

Uncertainties in Aquatic Modeling and Monitoring Estimates

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

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

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

The assessment relies on maximum use patterns (**APPENDIX 1-2**). In situations where use patterns are less than the labeled maximums, environmental exposures will be lower.

The aquatic modeling conservatively assumes that the waterbody abuts the treated area. As such, any reduction in loading from runoff that could occur as the result of managed vegetative filter strips or unmanaged naturally occurring interfaces between treated areas and waterbodies are not considered.

The aquatic modeling assumes a constant wind of 10 mph blowing directly toward the waterbody (**Section 3.4.2. Spray Drift**). These assumptions are conducive to drift transport and result in maximum potential loading to the waterbody. However, in many situations the wind will not be blowing constantly and directly toward the water body at this speed; therefore, aquatic deposition will likely be less than predicted. Additionally, many labels and applicator best management practices encourage not applying pesticides when the wind is blowing in the direction of sensitive areas (*i.e.*, listed species habitat). Lastly, reductions in spray drift deposition due to air turbulence, interception of spray drift on nearby plant canopy, and applications during low wind speeds are not considered in the spray drift estimates; therefore, loading due to spray drift may be over-estimated.

There is uncertainty associated with the selection of PWC input parameters. In this regard, one of the important parameters that can impact concentration estimates is the selection of application dates (**Section 3.4.3. Application Timing**); runoff and potential pesticide loading are greatest when applications immediately precede major precipitation events. Although the pesticide application dates are selected to be appropriate and protective (*i.e.*, selected with consideration for label restrictions and simulated cropping dates, pest pressures, and high precipitation meteorological conditions), uncertainty nevertheless results because the application window (the time span during a season that a pesticide may likely be applied) for a pesticide may be wide and actual application dates may vary over the landscape. While data sources exist that allow for determination of historical application dates (*e.g.*, California’s Pesticide Use Report and pesticide use surveys), it is uncertain how these dates reflect future application events. Additionally, the PWC model uses the same application dates for the 30-year simulation. While it is unlikely that an application would occur on the same dates every year for 30 years, this modeling process allows for a distribution of EECs to be developed that captures the peak loading events.

The PWC is a field-scale model. Flowing water bodies such as streams and rivers with physical parameters consistent with aquatic bins 3 and 4 have watershed areas well beyond those of typical agricultural fields. Initial modeling efforts in previous BEs, applying the field-scale model to these large watersheds and using the same scenario parameters as those used for the other bins, resulted in extremely high EECs which have not been observed in the environment, nor would be expected to occur due to fluid dynamic processes such as advective dispersion (Figure 3-3), where the peak concentration is dampened as it moves from a low flowing stream (bin 2) to a higher flowing river (bins 3 and 4). It is acknowledged that a watershed/basin-scale model capable of evaluating the impact of pesticide and water transport at the field-scale and aggregating these loadings to waterbodies at the larger watershed-scale is needed to evaluate these flowing aquatic systems.



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

## Uncertainties the Plant Assessment Tool (PAT)

The PAT model does not account for site specific field management and hydrology (e.g., terracing, contour farming, runoff and erosion controls, irrigation/drainage ditches, rills and creeks) which may result in less opportunity for runoff into the T-PEZ. Many different factors (e.g., slope; surface roughness; flow path length; etc.) can influence the occurrence, distance of, and prevalence of runoff onto the T-PEZ. These factors may vary greatly between different application sites (e.g., corn; wheat; potato; grape; bare field; turf).

The PAT model assumes that the water leaving the field as surface runoff is driven primarily by the amount of rainfall and the curve number, which is a function of the land use (i.e. row crops, pasture, fallow), management (i.e., straight row cropping, conservation tillage, etc.), and hydrologic soil conditions (i.e., high runoff potential with very slow infiltration rates; Young and Fry, 2016). Runoff leaving the field is assumed to enter the T-PEZ along the downslope field edge and coverage of the T-PEZ area conceptually happens instantaneously as the calculations are on a daily timestep rather than shorter timestep (e.g., hourly). As a result, the T-PEZ does not account for differences in the runoff loading (e.g., point entry and fan shaped sheet flow vs. uniform sheet flow entry), gradients in concentration due to interception and infiltration (e.g., buffering capacity of the T-PEZ), rain intensity and infiltration capacity relationships (e.g., pulsed rain events vs. one intense rain event). These natural features of the landscape may result in higher concentrations from runoff at the edge of the T-PEZ nearer the treated field than estimated in the model.

There are many different types of wetlands (e.g., depressional, groundwater fed, flow through, permanently flooded, and ephemeral) that may be present in landscapes receiving runoff from pesticide use sites. The default WPEZ model was selected to be representative and conservative (in terms of final pesticide concentrations) and acts as a surrogate for other types of wetlands. This assumption may result in overestimation of pesticide loading and fate than would be observed in some wetland systems.

Measures of Terrestrial Exposure

Terrestrial animals may be exposed to glyphosate through multiple routes of exposure, including diet, drinking water, dermal and inhalation exposure. Terrestrial dietary items may consist of plants, invertebrates or vertebrates (amphibians, reptiles, birds or mammals) that inhabit terrestrial areas or aquatic dietary items (fish, invertebrates or plants). However, due to glyphosate’s log Kow value (<0.001), significant bioaccumulation in aquatic food items is not expected and potential risk from this route of exposure is considered low. Therefore, estimates of exposure through consumption of aquatic food items using KABAM or BCF values are not calculated. A detailed discussion of the conceptual framework for estimating terrestrial exposure concentrations is provided in **ATTACHMENT 1-1**.

Two major parameters are used in terrestrial exposure modeling to characterize a species: body weight and diet. Estimates of body weights are necessary to estimate dose-based exposures through diet, drinking water, inhalation and dermal exposure routes. Information on the dietary requirements of listed species are necessary to determine relevant exposures through consumption of contaminated prey. Species-specific assumptions related to diet and body weight are provided within the model. The foliar dissipation half-life of the chemical can also impact the duration of exposure to predicted terrestrial EECs.

Two major parameters are used in terrestrial exposure modeling to characterize a species: body weight and diet. Estimates of body weights are necessary to estimate dose-based exposures through diet, drinking water, inhalation and dermal exposure routes. Information on the dietary requirements of listed species are necessary to determine relevant exposures through consumption of contaminated prey. Species-specific assumptions related to diet and body weight are provided within the model. The foliar dissipation half-life of the chemical can also impact the duration of exposure to predicted terrestrial EECs. Foliar dissipation data were available for glyphosate use on alfalfa (2 crop residue trials; MRID 45646001) in which the foliar dissipation half-lives for the two trials were 4.1 and 7.5 days. Additionally, in the paper by Willis and McDowell (1987), the reported foliar dissipation half-life based on forest foliage, was 14.4 days. Furthermore, in another paper by Feng and Thompson (1989), foliar dissipation on forestry brush foliage on two shrubs was examined and the calculated (by reviewer) foliar dissipation half-life was 4.9 days for each plant. These five values were used to calculate a 90th percentile upper confidence limit foliar dissipation half-life of 12 days. This foliar dissipation half-life of 12 days is used for glyphosate for terrestrial exposure modeling.

To improve efficiency and expand EFED’s modeling capabilities to other, non-dietary routes of exposure for terrestrial organisms, models have been developed to integrate the relevant exposure pathways and allow for batch processing of multiple analyses. For use in the pilot BEs (USEPA, 2016f), the Terrestrial Effects Determination (TED) tool was developed, which integrated T-REX, T-HERPS, the earthworm fugacity model, components of KABAM and AgDRIFT into one model platform. As part of the development of the draft biological opinion for chlorpyrifos, diazinon and malathion, the TED tool was converted to the terrestrial MAGtool, which further expanded the TED tool to predict the magnitude of effect at a population scale and incorporate the degree of overlap of a species range with potential use sites for a chemical (and associated off site transport areas) into the effects determination. The MAGtool has replaced the TED tool for modeling terrestrial exposure in the biological evaluations. A complete description of the MAGtool can be found in **ATTACHMENT 4-1**.

When the MAGtool is run for each species, terrestrial exposure concentrations are uniquely calculated for each species depending on relevant use overlap with the species range, available usage data, application rates associated with these relevant uses and the dietary items, habitat and obligate relationships for that species. As EECs will vary for each species, they are reported with the individual species in the individual effects determinations (**APPENDIX 4-1**).

To provide a bounding of potential terrestrial EECs used in the effects determinations, EECS were calculated for the range of application rates for glyphosate (a lower bound application rate of 0.28 lb a.i./A with 1 application per year and an upper bound application rate of 40 lb a.i./A with 1 application per year) and are provided below in Table 3-10. How the EECs will be applied will vary with each step of the analysis (*e.g.,* use of upper bound EECs in Step 1 vs. distribution of EECs in Step 2) and could be slightly higher with mid-range application rates applied multiple times. Additionally, other information considered in Step 2 (*e.g.,* typical use rates, use rates based on maximum usage in a species range, distribution of EECs, etc.), could alter the EECs used to assess a species exposure. All uses for glyphosate and associated application rates are provided in **APPENDIX 1-2**. Table 3-10summarizes the mean and upper bound dietary-based EECs and the associated base model that is used in the MAGtool to predict the EECs. Glyphosate uses also include granular formulations; these are analyzed separately and are discussed in **APPENDIX 4-5**.

Table 3-10. Mean and Upper Bound Dietary Based EECs Calculated for Food Items Consumed by Listed Mammals, Birds, Terrestrial-phase Amphibians or Reptiles Based on Foliar Applications. Values represent potential exposures for animals feeding on the treated field or in adjacent habitat directly adjacent to the field.

|  |  |  |  |
| --- | --- | --- | --- |
| **Food Item** | **Base Model** | **Lower bound application rate****(0.28 lb a.i./A x 1 application/year)** | **Upper bound application rate****(40 lb a.i./A x 1 applications/year)** |
| **Upper Bound** | **Mean** | **Upper Bound** | **Mean** |
| Short Grass | T-REX | 67.2 | 23.8 | 9600 | 3400 |
| Tall Grass, nectar and pollen | T-REX | 30.8 | 10.08 | 4400 | 1440 |
| Broadleaf plants | T-REX | 37.8 | 12.6 | 5400 | 1800 |
| Seeds, fruit and pods | T-REX | 4.2 | 1.96 | 600 | 280 |
| Arthropods (above ground) | T-REX | 26.3 | 18.2 | 3760 | 2600 |
| Soil-dwelling invertebrates (earthworms) | Earthworm fugacity | 4.4 | NA | 633 | NA |
| Small mammals (15 g, short grass diet) | T-HERPS | 64 | 22.7 | 9153 | 3242 |
| Large mammals (1000 g, short grass diet)2 | T-HERPS | 10.3 | 3.63 | 1467 | 519 |
| Small birds (20 g, insect diet) | T-HERPS | 30 | 20.7 | 4282 | 2961 |
| Small terrestrial phase amphibians or reptiles (2 g; insect diet) | T-HERPS | 1.5 | 1 | 209 | 144 |
| Fish, aquatic invertebrates and aquatic plants | NA3 |  |

1 NA as upper bound and mean residues only applicable to items dependent on residues on foliage

2 Also represent residues in carrion.

3 NA due to lack of bioaccumulation of glyphosate in aquatic dietary items

Literature Cited

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

Al-Rajab, Abdul Jabbar and Michael Schiavon. 2010. Degradation of 14C-glyphosate and aminomethylphosphonic acid (AMPA) in three agricultural soils. Journal of Agricultural Sciences. 22(9):1374-1380.

Baker, N.T., Stone, W.W., Wilson, J.T., and Meyer, M.T., 2006. Occurrence and Transport of Agricultural Chemicals in Leary Weber Ditch Basin, Hancock County, Indiana, 2003-2004. 2005 National Water Quality Assessment Program. U.S. Department of the Interior, U.S. Geological Survey, Scientific Investigations Report, 2006

Battaglin, William A, Rice, Karen C, Focazio, Michael J, Salmons, Sue, and Barry, Robert X. The occurrence of glyphosate, atrazine, and other pesticides in vernal pools and adjacent streams in Washington, DC, Maryland, Iowa, and Wyoming, 2005 – 2006. Environ Monit Assess (2009). 155: 281-307.

Bromilow, R., G. Briggs, M. Williams, J. Smelt, L. Tuinstra, W. Traag. 1986. The Role of Ferrous Ions in the Rapid Degradation of Oxamyl, Methomyl and Aldicarb in Anaerobic Soils. *Pestic. Sci*., 17:535-547

Chapman and Cole, 1982. “Observations on the influence of water and soil pH on the persistence of insecticides.” J. Environ Sci Health B. 1982; 17(5): 487-504

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

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

Majewski, M. S., & Capel, P. D. 1995. Pesticides in the Atmosphere: Distribution, Trends, and Governing Factors. Chelsea, MI: Ann Arbor Press.

Quaghebeur, D., De Smet B., De Wulf, E., Steurbaut, W., 2004. Pesticides in rainwater in Flanders, Belgium: results from the pesticide monitoring program 1997-2001. Journal of Environmental Monitoring 6: 182-190

Sargeant, D, Dugger, D. Newell, E., Anderson, P, Cowles, J. Surface Water Monitoring Program for Pesticides in Salmonid-Bearing Streams 2006-2008 Triennial Report, February 2010 (Washington State Department of Ecology and Washington State Department of Agriculture) https://fortress.wa.gov/ecy/publications/summarypages/1003008.html; <http://agr.wa.gov/PestFert/natresources/docs/swm/2008_swm_report.pdf>

Sargeant, D., Newell, E., Anderson, P., Cook, A. Surface Water Monitoring Program for Pesticides in Salmonid-Bearing Streams 2009-2011 Triennial Report, February 2013 (Washington State Department of Ecology and Washington State Department of Agriculture) <http://agr.wa.gov/FP/Pubs/docs/377-SWM2009-11Report.pdf>

Smelt, J. H., A. Dekker, M. Leistra, and N. Houx. 1983. Conversion of Four Carbamoyloximes in Soil Samples from Above and Below the Soil Water Table. *Pestic. Sci.* 1983, 14(2):173-181.

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

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

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

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

Willis, G.H. and McDowell, L.L. 1987. Pesticide persistence on foliage. Environ. Contam. Toxicol, 100:23 73.

WU, X., Sun, X., Zhang, C., Gong, C., and Hu, J. Micro-mechanism and rate constants for OH-initiated degradation of methomyl in Atmosphere. Chemosphere 107:331-335.

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

1. The exposure models can be found at: <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment> [↑](#footnote-ref-2)
2. Bush berries like raspberries can live 15 to 20 years before replanting, bush berries like blueberries can live 40 to 50 years, and citrus and vineyards can last for 20 years before replanting. [↑](#footnote-ref-3)
3. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling> (accessed January 2020) [↑](#footnote-ref-4)
4. <https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values> (accessed January 2020) [↑](#footnote-ref-5)
5. The draft guidance is available at www.regulations.gov docket number: EPA-HQ-OPP-2013-0676 [↑](#footnote-ref-6)
6. <https://iaspub.epa.gov/waters10/attains_nation_cy.control?p_report_type=T> [↑](#footnote-ref-7)