**EXPOSURE CHARACTERIZATION for Diazinon**

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# Environmental Transport and Fate Characterization

Physical-chemical properties and dissipation-related parameters for diazinon and its major degradates of concern are provided in **Table 3-1**. A more complete discussion of the fate information is available in **APPENDIX 3-1 Environmental Transport and Fate Data Analysis**. Data summarized here include data submitted to the U.S. EPA and open literature data including ECOTOX studies classified as ECOTOX plus. Open literature data were included when the information was determined to add to the overall understanding of the environmental fate of diazinon and diazoxon. **APPENDIX 3-2 Diazinon Fate Open Literature** summarizes the listing of ECOTOX plus studies listed as fate related.

Diazinon enters the environment via direct spray and spray drift onto soil, foliage, and/or water. The environmental fate properties of diazinon along with monitoring data identifying its presence in surface waters, air, and in precipitation indicate that important transport pathways include runoff and spray drift.  Volatilization, atmospheric transport, and subsequent deposition of diazinon to aquatic and terrestrial habitats also occur.

Based on diazinon’s aerobic soil metabolism and aerobic and anaerobic aquatic metabolism data, diazinon is not considered persistent[[1]](#footnote-2) in the environment, with half-lives on the order of days to weeks (representative[[2]](#footnote-3) half-life values range from 9 to 57 days).  Diazinon also degrades via hydrolysis with time to 50% decline (DT50) values of 2 days at pH 4, 12 days at pH 5, and ranging from 62 to 139 days at pH 7 and 9. Hydrolysis half-lives indicate that diazinon is classified as persistent in aquatic and terrestrial environments where microbial activity is not present; however, microbial activity is expected in most natural environments.  The dominant degradation process is expected to depend on environmental conditions.  At low pH (pH 4 to 5), hydrolysis may be the primary degradation process, while at higher pH (above pH 5), aerobic metabolism will be more important.  Terrestrial field dissipation DT50s range from five to 20 days in 18 field studies, and do not exhibit any obvious relationship with formulation. There is no obvious relationship between DT50 values and cropped versus bare fields.  Residues of diazinon were still present in soils after one year with repeated application at some sites, but not at others. Results from the terrestrial field dissipation studies are consistent with those observed in the lab. Dissipation DT50 values range from 5 to 20 days and aerobic soil metabolism DT50 values range from 9 to 57 days. The presence of residues in terrestrial field dissipation studies is consistent with laboratory studies when DT90 values and the shape of the decline curves are considered. Aerobic soil DT90 values range from 28 to 188 days for diazinon and 28 to 1285 days for diazinon plus lost radioactivity[[3]](#footnote-4) and some degradation curves showed an initial rapid rate of decline followed by more gradual decline. Diazinon does undergo atmospheric degradation; the half-life estimated for the average 12-hour day time concentration of hydroxyl radicals (1.5×106 molecules (radicals/cm3) in the troposphere at 40oC (104oF) is 1.3 hours.

Diazinon is classified as moderately mobile to slightly mobile (KOCs range from 138-3779 L/kg)[[4]](#footnote-5) and has the potential to reach surface water through runoff and soil erosion. Overall, soil/sediment-water distribution coefficients increase with increasing percent organic-carbon. Diazinon has the potential to reach groundwater especially in high-permeability soils with low organic-carbon content and/or the presence of shallow groundwater. The maximum depth of leaching in the terrestrial field dissipation studies is 48 inches.  In water and sediment, diazinon will be present both in the water column and bound to sediments. Based on measured octanol-water partition coefficients (Kows) and KOCs, exposure to sediment-dwelling organisms is likely to occur. Diazinon is semivolatile and may also be transported in air in both the vapor form and associated with particles. Diazinon is oxidized to diazoxon by hydroxyl radicals and ozone. Based on the drinking water treatment data (Acero *et al.*, 2008; Beduk *et al.*, 2011; Chamberlain *et al.*, 2012; Duirk *et al.*, 2009; Magara *et al.*, 1994; Ohashi *et al.*, 1994; Wu *et al.*, 2009; Zhang and Pehkonen, 1999), it is possible that diazoxon could form in air in the presence of ozone.

Empirical bioconcentration factors (BCF) for diazinon range from 3 to 82 L/kg-wet weight aquatic invertebrates and 18 to 213 L/kg-wet weight in fish, and the estimated time to steady state for diazinon is ≥4 days (estimated using KABAM). Based on diazinon’s log octanol-water partition coefficient (kow), it is possible that mammals and birds could be exposed to diazinon via consumption of aquatic animals exposed to diazinon in water.  Based on diazinon’s log Kow and its log octanol-air partition coefficient (log KOA) of 8.4, diazinon is likely to bioconcentrate in terrestrial organisms, if it does not degrade and is not metabolized (Armitage and Gobas, 2007; Gobas *et al.*, 2003; USEPA, 2008, 2009b). However, the short atmospheric half-life of diazinon will limit the amount of diazinon in air over time and metabolism of diazinon is known to occur in vertebrates.

The only identified degradate of concern[[5]](#footnote-6) for diazinon is diazoxon. Diazoxon has been identified as a residue of concern for both human health and ecological risk assessments. Diazoxon was observed in only one submitted aerobic soil metabolism study at a maximum of 0.6% applied radioactivity and in an air photolysis study where it formed before the photolysis portion of the study began. Limits of quantitation for diazoxon were high (0.01 to 0.02 mg/kg-soil) in the studies where it was examined and there was a portion of unidentified residues in submitted laboratory studies. Although formation and degradation of diazoxon cannot be quantified from available laboratory fate studies involving diazinon, diazoxon has been detected in air, rain, fog (Majewski and Capel, 1995) and surface waters in the United States (USGS, 2011). Organophosphates that contain a phosphothionate group (P=S) such as diazinon are known to transform to the corresponding oxon analogue containing a phosphorus-oxygen double bond (P=O) instead. This transformation occurs via oxidative desulfonation and can occur through photolysis and aerobic metabolism, as well as other oxidative processes. Disinfection with chlorine or ozone converts diazinon to diazoxon (Acero *et al.*, 2008; Beduk *et al.*, 2011; Chamberlain *et al.*, 2012; Duirk *et al.*, 2009; Magara *et al.*, 1994; Ohashi *et al.*, 1994; Wu *et al.*, 2009; Zhang and Pehkonen, 1999) and similar reactions with ozone could occur in the natural environment. In surface water monitoring data wherein residues of both diazinon and diazoxon were detected, the ratios of the concentrations of diazoxon to diazinon ranged from 0 to 0.5. The atmospheric degradation half-life estimated for the average 12-hour day time concentration of hydroxyl radicals at 30oC (104oF) was 4.1 hours (MRID 49049902).

Table 3-1. Physical/chemical and environmental fate properties of diazinon and diazoxona

| **Chemical Fate/Parameter** | **Range of Values (Number of Values)** | | |
| --- | --- | --- | --- |
| Common name | Diazinon | | Diazoxon |
| Structure | Chemical structure for DIAZINON | | 2D chemical structure of 962-58-3 |
| IUPAC Name | *O*,*O*-diethyl *O*-2-isopropyl-6-methylpyrimidin-4-yl phosphorothioate | | Phosphoric acid, diethyl 6-methyl-2-(1-methylethyl)-4-pyrimidinyl ester |
| Chemical Formula | C12H21N2O3PS | | C12H21N2O4P |
| Molecular Mass (g/mole) | 304.35 | | 288.28 |
| Vapor Pressure (Torr, 25°C) | 7.22×10-5  6.6×10-5 | | 1.1×10-5estimated |
| Solubility (25°C) (mg/L) | 59.5 pH 6.07  65.5 pH not reported | | 245 estimated pH NR |
| Octanol-water partition coefficient (Kow) | 4898 (log KOW=3.69) at 24oC  6393 (log KOW=3.8) at 25oC | | 117 (log KOW 2.07) estimated |
| Atmospheric Degradation half-life (hours, 39.5oN) | 1.3 at 40oC  In presence of OH radicals | | 4.1 30oC  In presence of OH radicals |
| Hydrolysis half-life (days) | pH 4, 25oC | 1.93 | No data |
| pH 5, 23-25oC | 12.4 |
| pH 7, 25oC | 82.3 |
| pH 7, 23-25oC | 139 |
| pH 9, 23-25oC | 77.1 |
| pH 9, 25oC | 61.9 |
| Aqueous photolysis half-life (days) | Stable at pH 7, 25oC  0.3-1 in presence of NO3-, CO32-, and DOC at summertime ambient light 53oN, pH 7, 20oCe | | No data |
| Soil photolysis half-life (days) | No acceptable data | | No data |
| Aerobic soil metabolism representative half-lifea range (days) | 9 – 57 (5)  Not Persistentb | | No data |
| Anaerobic soil metabolism representative half-lifea range (days) | No data | | No data |
| Aerobic aquatic metabolism representative half-lifea range (days) | 10 – 16 (2) in water-soil  6.3 – 41.0 (4) in surface waterf | | No data |
| Anaerobic aquatic metabolism representative half-lifea range (days) | 24.5 (1) | | No data |
| Organic-carbon normalized soil –water distribution coefficients (Koc) L/kg-OC | 138-3779 (32)d  Moderately to Slightly Mobilec | | 174.7 (estimated using EPIWeb 4.1)  Moderately Mobilec |
| Terrestrial field dissipation DT50s | 5 – 20 (18) | | NR |
| Aquatic field dissipation DT50s | Not available | | No data |
| Bioconcentration factor (L/kg-wet weight) | 3 – 82 (8) in aquatic invertebrates  18 – 213 (15) in fish | | No data |

NR=not reported; DT50=time to 50% decline of residues

a Representative half-life values are generated using the North American Free Trade Agreement (NAFTA) guidance for calculating degradation kinetics (NAFTA, 2012; USEPA, 2012b). The representative half-life considers both initial and later (potentially slower) portions of the decline curve and is not necessarily numerically similar to the value of the DT50 , rather it provides an input value for modeling that is generally expected to be conservative. The actual DT50 and DT90 from the representative degradation kinetic equations for the curve are used for descriptive purposes and for understanding the decline curve and the nature of the representative half-life used in modeling, see **APPENDIX 3-1 Environmental Transport and Fate Data Analysis** for these values.

b Based on the Toxic Release Inventory classification system where half-lives greater than 60 days in water, soil, and sediment are considered persistent and half-life greater than six months are considered very persistent (USEPA, 2012a).

c Mobility was classified using the Food and Agriculture Organization (FAO) classification system (FAO, 2000).

d Sorption coefficients for diazinon did not have reliable mass balances, did not confirm equilibrium was achieved, did not account for sorption to test systems, did not confirm the identity of the compound associated with radioactivity, or had added solvents in the test systems and are considered supplemental. The EPIWeb estimated sorption coefficient is within the range of measured values.

e (Ukpebor and Halsall, 2012)

f (Bondarenko *et al.*, 2004)

Three field dissipation studies (MRID 41490401, 41490402, and 41490403) were conducted in which diazinon was applied to apple orchards in Adams County, Pennsylvania six times at 3 lbs a.i./A[[6]](#footnote-7) and concentrations of diazinon were measured in an adjacent pond[[7]](#footnote-8). The final applications took place in July. The maximum and range of detected concentrations in surface water are summarized in **Table 3-2**. Samples were collected immediately after the first applications and at irregular intervals over the study period (*e.g.,* sampling intervals ranged from daily to weekly). Diazinon was detected shortly after the applications and rainfall events with concentrations decreasing through October, when the final samples were collected. The final mean measured concentrations[[8]](#footnote-9) ranged from 0.2 to 0.5 µg/L. The Jack Ely and Ronald Rice sites are similar to the aquatic bin 7 (high volume static bin). The R.R. Showers site is between aquatic bin 6 (moderate volume static bin) and 7.

Table 3-2. Summary of diazinon concentrations in ponds near apple orchards after applications of diazinon.\*

| Site/MRID | Maximum Diazinon Concentration (µg/L) | Range of Average Concentrations+  µg/L | Area of Watershed  Pond Volume, Surface area | Comments |
| --- | --- | --- | --- | --- |
| Jack Ely/ MRID 41490402 | 82.1 | 0.5 to 44.1 | 10.2 acres watershed  10.2 acres treated  1.7 acre pond, 8.3 acre-feet | Detectable residues in pond sediment |
| R.R.Showers/ MRID 41490403 | 12.8 | 0.5 to 9.2 | 69.4 acre watershed  24.2 acres treated  4.9 acre pond, 21.5 acre-feet |  |
| Ronald Rice /MRID 41490401 | 113.0 | 0.6 to 53.4 | 33.7 acres watershed  14.1 acres treated  0.7 acre pond, 3.2 acre-feet | Stream in same watershed as the pond. Residues not quantifiable in pond sediment due to unacceptable recoveries in fortified samples. |

\*These studies were classified as supplemental.

+ Average of individual samples collected from three different zones of the pond on the same day.

Szeto *et al.* (1990) evaluated concentrations of diazinon and diazoxon in cranberry bogs and adjacent waters after application of Diazinon 5G (a granular formulation)[[9]](#footnote-10). Diazinon was applied at a rate of 6 kg a.i./ha (5.35 lbs a.i./A)[[10]](#footnote-11) by aircraft to 19 hectares of cranberries in nine beds on July 26 and August 8. Cranberries bogs were surrounded by irrigation ditches, reservoirs, and waterways linking to two small tributaries to the Fraser River (near Forth Langley, British Columbia). Cranberry bogs were irrigated in April and water held with gates until after harvest. Sediment and water was collected at six stations within plots and outside of plots. Stations were as follows:

* one in an irrigation ditch in treatment plot,
* one in the reservoir adjacent to treatment plot,
* two in waterways outside of the dyke, and
* one at each of the two tributaries approximately 100 m downstream from the edge of the treatment plot.

Samples were collected at 10 days before the first application, pre-spray, post-spray, and at intervals up to 137 days after application. Recoveries of diazinon and diazoxon from water were near 100% but recoveries from sediment (69 to 76%) were low, likely due to hydrolysis (sediment pH ranged from 4.4 to 6.0). The limit of detection was 0.1 µg/L for sediment and 10 µg/kg for sediment. Results for the waterways and tributaries were similar and the results were averaged in the report (**Table 3-3**). Diazoxon was not detected in any of the samples of water or sediment. The maximum diazinon concentration in water detected was 456 µg/L in irrigation ditches which decreased to below 100 µg/L within three to four days after treatment. Concentrations in the adjacent reservoir were lower with a maximum of 78.5 µg/L. Szeto *et al.* (1990) indicated residues observed in tributaries were much lower and were likely caused by leakage from the irrigation water through the gate between the reservoir and the waterways. Increased concentrations were also observed with a high rainfall event. Diazinon was also detected in sediment. Hydrolysis was likely a major loss mechanism. The pH of water ranged from 5.1 to 6.6, and diazinon is known to undergo hydrolysis in acidic environments and pH of sediment ranged from 4.4 to 6.0. This study was obtained from the open literature and the results are considered supplemental. **Table 3‑3** summarizes the concentrations of diazinon in water and sediment of cranberry bogs and adjacent waterways.

Table 3-3. Concentrations of diazinon in water and sediment of cranberry bogs and adjacent waterways

| Site | Max diazinon concentration in water (days after first app) in µg/L | | | Max diazinon concentration µg/kg-sediment (days after first app) | | |
| --- | --- | --- | --- | --- | --- | --- |
| After 1st App | After 2nd App | Final Detection | After 1st App | After 2nd App | Final Detection |
| Irrigation ditch | 338 (1d) | 456 (14d) | 0.2 (35 d) | 21200 (4d) | 8920 (21d) | 20 (137d) |
| Reservoir | 78.5 (2d) | 58.1 (17d) | 0.3 (51d) | 2380 (1d) | 110 (17d) | 10 (51d) |
| Waterways outside dyke | 29.1 (2d) | 2.6 (15d) | 0.1 (42d) | 80 (1d) | 20 (14d) | 10 (35d) |
| Tributaries 100 m downstream | 2.8 (4d) | 1.1 (15d) | 0.1 (35d) | 10 (4d) | Not detected | Not detected |

App= application

# Measures of Aquatic Exposure

In general, maximum application rates and minimum application retreatment intervals were modeled based on the Diazinon Use Summary Table (**APPENDIX 1-3 Use Summary Table**), unless otherwise noted.

## Aquatic Exposure Models

Aquatic exposures (surface water and benthic sediment pore water) were quantitatively estimated for diazinon for all currently registered uses by HUC 2 Regions (**Figure 3-1**) and by aquatic bin using the Pesticide Water Calculator version 1.38 (10/22/2015). Exposure estimates were also calculated for cranberries using PFAM version 1.0 (1/23/2013).

Diazinon specific modeling simulations were developed for each registered use, based on selection of PRZM5/VVWM[[11]](#footnote-12) scenarios and agronomic practices (*e.g.*, applications methods, dates) as described below. **APPENDIX 1-7 Diazinon Scenario Development** provides further information on all PRZM5/VVWM and PFAM model input values.

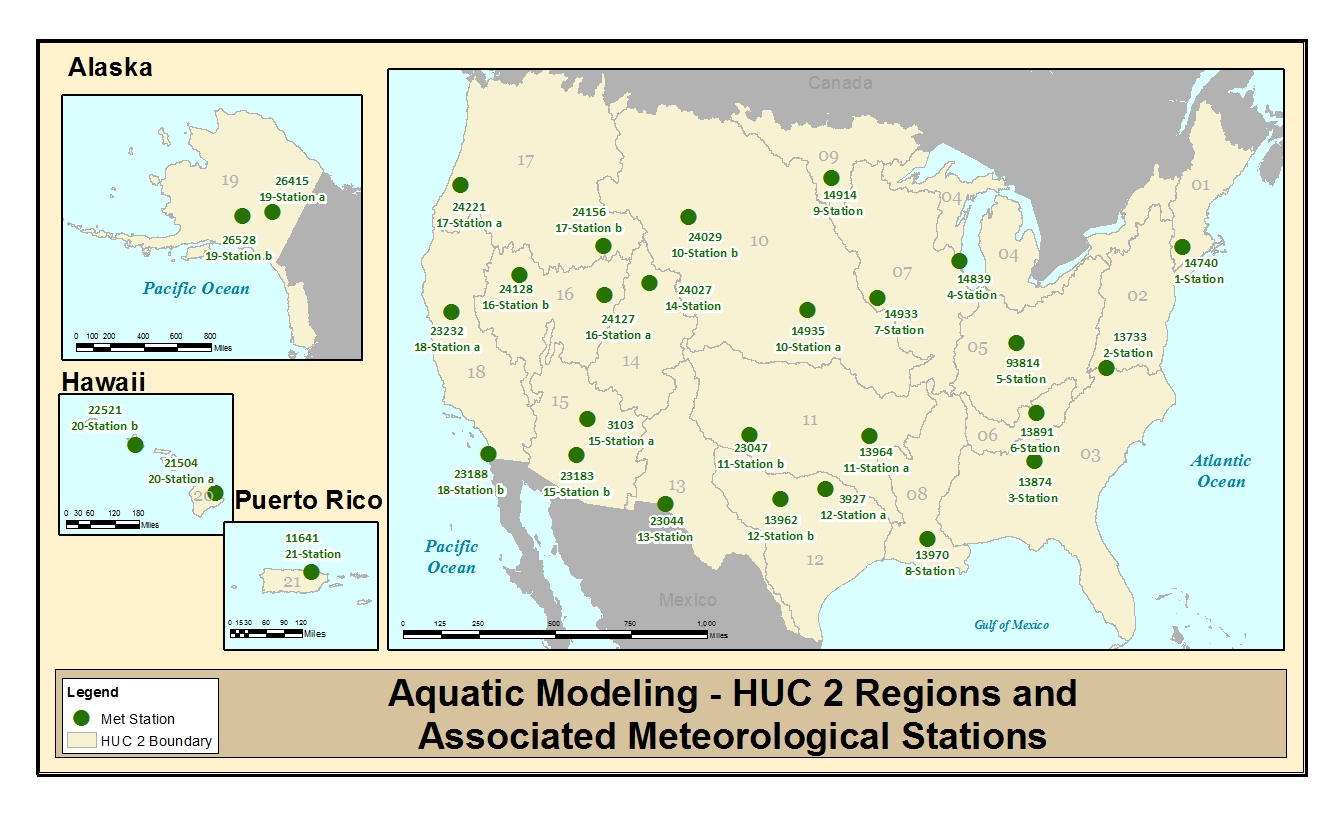


Figure 3-1. The 21 major hydrologic unit “Water Resource” regions (HUC-2) in the continental United States, overlain on State boundaries

## HUC and Use Site Crosswalk

Unless the label limits a use pattern to a particular geographic area (*e.g*., many uses are only allowed in Texas), the National Agricultural Statistics Census of Agriculture 2012 (NASS) data were used to determined which crops would be modeled for each HUC 2 region[[12]](#footnote-13). If the NASS data indicated any area of a crop grown (even if it was a small acreage) in a specific HUC 2 region, it was assumed that the crop was grown in that HUC 2 region and diazinon used on that crop within the HUC 2 region. If there were no reported NASS cropped acres grown in a particular HUC 2 region, then it was assumed that the use did not occur in the HUC. The results of this work are provided in the worksheet titled 'diazinon-NASS-HUC2 in **APPENDIX 3-4a Diazinon aquatic modeling inputs**.

## Scenario Selection

To generate spatially relevant aquatic exposure concentrations, PRZM5/VVWM-scenarios used in model simulations were selected based on the crop group HUC2 Region scenario matrix provided in the worksheet titled ‘scenario HUC’ in **APPENDIX 1-6 Use Site, General Land Cover Class, and HUC2 Matrix for diazinon**. An explanation of how the PRZM5/VVWM scenario matrix was developed is provided in **APPENDIX 1-7 Scenario Development**. The land cover and specific use site crosswalk is provided in **APPENDIX 1-6 Use Site, General Land Cover Class, and HUC2 Matrix for diazinon**.

## Application Practices

### Application Method

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

### Spray Drift

Diazinon formulations are all sprays, and the default spray drift inputs were assumed. The default spray drift fractions used in modeling are shown in **Table 3-4**. Some labels specify that applications must occur at a specified buffer distance (10 to 100 feet) to an aquatic area to prevent runoff into the waterbody; however, these spray drift buffer restrictions are not on all labels and only apply to ‘sensitive aquatic areas’. The impact of a 10-foot and 100-foot buffer on aquatic EECs is explored in the sensitive analysis; however, the language does not require that this buffer exists for all applications. Spray drift estimates for each bin, considering these buffers are described in **APPENDIX 3-3 Spray Drift Considerations for Diazinon**.

Table 3-4. Estimated spray drift fractions for different aquatic bins and application methods.1

| **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 | 0.32 | 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)** | 0.62 | 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 | 0.14 | 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 | 0.06 | 0.021 | 0.104 | 0.054 | 0.01 |
| Vineyard | 0.025 | 0.013 | 0.004 | 0.03 | 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 waterbody 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. | | | | | | | |

### Application Timing

In selecting application dates for aquatic modeling, EPA considers a number of factors including label directions, timing of pest pressure, meteorological conditions, and pre-harvest restriction intervals. Agronomic information was consulted to determine the timing of 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.

Diazinon 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 diazinon uses, PRZM5/VVWM model inputs for the application dates were chosen based on these timings, the crop emergence and harvest timings specified in the PRZM5/VVWM scenario, and precipitation data for the associated meteorological station. Application dates were selected to represent conservative and reasonable estimates. At plant applications were specified as occurring eight days before crop emergence. 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 PRZM5/VVWM scenario. When choosing an application date within a time window (*i.e*., dormant season or foliar application), the first or 15th of the month with the highest amount of precipitation (for the meteorological station for the PRZM5/VVWM scenario) for that time window was chosen. Pesticide loading to surface water is directly affected by precipitation events. Although efforts may be taken to avoid pesticide applications right before precipitation events, usage data are available showing that pesticide applications commonly occur a few days before or the day of 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.

## Special Agricultural Considerations

### Multiple Crop-cycles Per Year

The Diazinon Use Summary Table (see the worksheet titled ‘use summary’ in the diazinon-aquatic modeling work 10-20-15.xlsx’ excel file) indicates that many of the vegetable crops may have more than one crop planted on the same parcel of land in the same year. For example, the Diazinon Use Summary Table indicates that succulent beans may have one spring and one fall crop. When maximum annual application rates were specified for a vegetable crop in the Diazinon Use Summary Table, it was assumed that only one crop was planted per area of land per year for that crop, even when the registrant indicated more than one crop season per year could occur. This is because the maximum annual application rate cannot be exceeded regardless of the number of crops cycles per year.

When maximum application rates were specified on labels on a crop cycle basis and the Diazinon Use Summary Table indicated that multiple crop cycles per year would be allowed on the label, it was assumed that multiple crops per year could be planted on the same plot of land (**Table 3-5**). Multiple crop cycles per year were simulated for lettuce and for ornamentals grown in nurseries by simply assuming that the number of applications per year could be doubled or tripled. For use on Swiss chard, squash, and turnip, simulations where the same crop was planted on the same plot of land were not simulated; however, the exposure from the simulations for lettuce and nursery ornamentals, as well as, the additional simulations where different crops were rotated (discussed in the next paragraph) provide enough characterization to understand the potential exposure for these use patterns.

BEAD summarized some common crop combination scenarios for vegetable crops grown in four states where PRZM5/VVWM scenarios are readily available for vegetables (California, Florida, Texas, and Michigan) (**Table 3-6**). These application windows for crop combinations were used to further characterize potential exposure from planting more than one vegetable crop in the same year. For diazinon, many of the application parameters are the same across many vegetable crops that may be rotated.

Table 3-5. Diazinon uses with possible multiple crop cycles per year and maximum rates provided on a crop cycle basis1

|  |  |
| --- | --- |
| Crop Group | Crop cycles per year |
| Swiss Chard | 2 |
| Squash | 2 |
| Lettuce | 2 |
| Turnip | 3 |
| Nursery Ornamentals | 3 |

1 Based on the Diazinon Use Summary Table

Table 3-6. Multiple crop rotation simulations

| **State Simulated** | **HUC** | **Crops Simulated** | **Application Simulation** | | |
| --- | --- | --- | --- | --- | --- |
|  | **Crop 1** | **Crop 2** | **Crop 3** |
| California | 18 | Spinach, Cauliflower, Lettuce | At plant on 1/1 at 4.5 kg/ha, incorporate 2 inches | At plant 3/20 at 4.5 kg/ha with no incorporation | At plant 7/31 at 2.24 kg/ha with 2 inch incorporation; Foliar aerial on 8/30 at 0.6 kg/ha |
| Florida | 3 | Radish, Cabbage, Lettuce | At plant 10/1 at 4.5 kg/ha incorporate 2 inches | 11/18 ground at 4.5 kg/ha incorporate 2 inches | At plant 4/10 at 2.24 kg/ha with 2 inch incorporation; foliar aerial on 5/10 at 0.6 kg/ha |
| Michigan | 7 | Cabbage, Melon | 4/8 ground at 4.5 kg/ha incorporate 2 inches | At plant 7/21 at 4.5 kg/ha and incorporate 2 inches. | -- |
| Texas | 12 | Carrot, onion | 7/1 ground at 4.5 kg/ha incorporate 2 inches | At plant 10/15 at 4.5 kg/ha incorporate 3 inches | -- |

### Cranberry Modeling for Surface Water

The Pesticides in Flooded Applications Model (PFAM, version 1.00) and the PRZM5/VVWM are both used to estimate EECs for cranberry use. Some cranberries are grown in bogs, where the field is temporarily flooded to control pests, prevent freezing, and to facilitate harvest. After flooding, water may be held in a holding system, recirculated to other cranberry growing areas, or released to adjacent waterbodies (rivers, streams, lakes, or bays). The potential exposure from movement of residues in water released from bogs is evaluated using PFAM. PFAM estimates post-application residues in water that is introduced to the field and not mixed with any additional un-treated water. Diazinon concentrations in adjacent waters are expected to be lower than those estimated in the cranberry bog as diazinon will degrade and water will likely be diluted with uncontaminated water from other sources in the adjacent waterways. The extent of this reduction in concentrations depends on 1) the length of time the compound is in the water, 2) the distance the water will travel, 3) the amount of dilution and 4) whether the water it is mixed with also carries residues of diazinon. The PFAM model simulates application of pesticide to a wet or dry field and degradation in soil and/or water. If the pesticide is applied to dry soil, water may then be introduced into the field and movement of pesticide from soil into the water after application may occur. While the PFAM model does have the capability of simulating release of cranberry bog water into a well-mixed waterbody, this scenario was not simulated because a conceptual model (*i.e*., cranberry area, watershed area, mixing cell flow rate, and mixing cell size) is not currently available. The results from the PFAM modeling for cranberries are used as a line of evidence along with the PRZM5/VVWM results for the aquatic bins in evaluating the potential risk from the use of diazinon on cranberries.

#### PFAM

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, and allows for more flexibility in refinement of assessments when needed.

A 12-inch flood was modeled on October 1, followed by draining the bog on October 4th. A winter flood was also simulated. The modeled flood date was selected as a plausible date of harvest. Crop stages were estimated. The maximum aerial coverage for berry crops used in the OR berries PRZM scenario for the PRZM5/VVWM was used in PFAM as well. **Table 3-7** and **Table 3-8** summarize the PFAM inputs assumed for setting up the scenario and the PFAM inputs specific to diazinon. If the input is not listed, the default input was used in modeling. Release of water into a receiving water body was not simulated because a conceptual model for this is not currently available. Therefore, the bog EECs estimated using PFAM are used as a conservative estimate of exposure in the receiving water body.

Table 3-7. PFAM inputs specific to diazinon

|  |  |  |  |
| --- | --- | --- | --- |
| **Input Parameter** | **Value** | **Source** | **Comment** |
| **Chemical Tab, see Table 3‑9** | | | |
| **Applications Tab** | | | |
| Application rate | 3.0 lbs a.i./A  3.4 kg a.i./ha | Diazinon Use Summary Table |  |
| Number of Applications | 3 | --- | --- |
| Application dates | 07/18  8/1  8/15 | --- | Registrant (email to Khue Ngyuyen on 9/26/2014) and BEAD indicated in an email that the last day of application of diazinon to cranberry bogs would be in mid-August. Agricultural extension information is consistent with this timing. The minimum retreatment interval is 14 days. |
| Slow Release 1/day | 0 | -- | Not applicable |
| Drift Application | 0 | -- | Drift to an adjacent water body or mixing cell was not modeled. |
| **Flood Tab** | | | |
| Number of Flood Events | 5 | -- | Harvest occurs between September and November. Field is flooded just prior to harvest. Field may also be flooded over the winter from December through March 15 (Cape Cod Cranberry Growers Association, 2001). The winter flood height was assumed to be similar to the harvest flood height. In some areas, there is also a late water flood to control spring frost where the bog is flooded in late April for one month. This was not simulated. |
| Date of Event 1 (Month-Day) | 01-01 | -- |
| Turn Over (1/day) | 0 | Assumed |
| **Days After (Month-day)** | **Fill Level, Min Level (m)** | **Weir (m)** |
| 0 (Jan -1) | 0.305 | 0.458 |
| 75 (Mar 17) | 0 | 0 |
| 273 (Oct-1) | 0.305 | 0.458 |
| 276 (Oct-4) | 0 | 0 |
| 334 (Dec-1) | 0.305 | 0.458 |

Table 3‑8. Summary of model inputs for the crop and physical tab input sheets in PFAM

|  |  |  |
| --- | --- | --- |
| **Crop Tab** | | |
| Zero Height Reference | 05/01 | Information from Maine Cooperative Extension (Armstrong, 2015) |
| Days from Zero Height to Full Height | 120 (08/29) | Assumed |
| Days from Zero Height to Removal | 153 (10/1) | Assumed |
| Maximum Fractional Areal Coverage | 0.2 | Value from OR berries PE scenario |
| **Location Tab** | | |
| Meteorological files | CT W14740  NJ W14734  WIa W14839  WIb W14920  OR W24221 | Weather stations from cranberry growing areas |
| Latitude | 42.3 | Latitudes are CT 41.6, 40.0 NJ, 44.5 in WI, and 44.0 in Oregon. These are close enough that a default latitude was chosen. |

#### PRZM5/VVWM

PRZM5/VVWM were used to model applications of diazinon to cranberries in a terrestrial environment. Cranberries are typically grown in bogs, and the direct surface water runoff calculations in the PRZM5/VVWM were not designed to represent this sort of circumstance. An analysis of runoff related to cranberry bogs in the New England area also describes some unique hydrology considerations for cranberry bogs,

“*Glaciation and the distribution of glacial deposits greatly influence the hydrologic characteristics of southern New England streams and rivers. Bog sites in low-lying southeastern coastal areas of Massachusetts and Rhode Island have significant areas (greater than 50%) of stratified sand and gravel glacial deposits and floodplain alluvium deposits. These stratified deposits are conducive to high infiltration rates, large storage capacities, and significant baseflow contributions to the surface water channels. The combination of stratified, highly conductive deposits and the low topographic relief allows water to move through the subsurface between surface water basins. Hence, peak discharges cannot be accurately computed by procedures based on direct surface runoff alone*.” (USDA, 2012)

While the typical surface runoff simulated in the PRZM5/VVWM does not apply to cranberries grown in bogs because of the unique hydrology of cranberry bogs, residues related to runoff from cranberries will occur and the PRZM5/VVWM is the tool available to capture exposure due to transport in runoff and spray drift. Run-off from cranberry bogs may occur through subsurface flow and when cranberries are not grown in flooded bogs. Some cranberries are dry harvested (on both the east and west coast, and Wisconsin) and are not grown in a depressed area or in these hydrologically unique areas. Therefore, the PRZM5/VVWM was also used to calculate EECs for cranberries and both runoff and spray drift were simulated. Regionally-specific meteorological data and conditions were used in conjunction with the PRZM5/VVWM to represent those parts of the country where cranberries are grown.

## Non-Agricultural Uses and Considerations

There are no non-agricultural uses, such as residential or right-or-way applications, approved for diazinon.

## Aquatic Modeling Input Parameters

For Step 1 analysis, aquatic exposure modeling was not performed, as the action area for diazinon encompasses the entire United States. The following sections discuss methods used for modeling under Step 2. Complete results of the Step 2 analysis are provided with the release of the complete BE. Summaries of the model input parameter values used in the PRZM5/VVWM and PFAM are presented in **Table 3-9**. Input parameters were selected in accordance with EFED’s following guidance documents:

* Guidance for *Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides*, Version 2.1[[13]](#footnote-14) (USEPA, 2009),
* *Guidance for Evaluating and Calculating Degradation Kinetics in Environmental Media[[14]](#footnote-15)* (NAFTA, 2012; USEPA, 2012c), and
* *Guidance on Modeling Offsite Deposition of Pesticides Via Spray Drift for Ecological and Drinking Water Assessmen*t[[15]](#footnote-16) (USEPA, 2013),

Table 3-9. Input values used for Tier II surface water modeling with PRZM5/VVWM and PFAM (Chemical Tab Sheet).

| **Parameter (units)** | **Residue** | **Value (s)** | **Source** | **Comments** |
| --- | --- | --- | --- | --- |
| Organic-carbon Normalized Soil-water Distribution Coefficient (KOC (L/kg-OC)) | Diazinon | 824  138, 3779 for characterization | EPIWEB v4.1  (Arienzo *et al.*, 1994; IglesiasJimenez *et al.*, 1996; Nemeth-Konda *et al.*, 2002) | Available measured KOC values were considered supplemental and range from 138 to 3779 L/kg-OC. Across sorption studies, Kd values correlated with the percent organic carbon. The average value (824 L/kg) was used to determine EECs used in the main risk characterization. The minimum (138) and maximum (3779) measured values were also used to characterize uncertainty. As the measured values are all supplemental with varying degrees of reliability. |
| Water Column Metabolism Half-life or Aerobic Aquatic Metabolism Half-life (days) and Reference Temperature | Diazinon | 13.2 at 25oC | MRID 46386604 | Represents the 90 percent upper confidence bound on the mean of two representative half-life values. Values were adjusted to 25oC because the studies were conducted at two different temperatures. The lowest (9.94) and highest (16.3) value measured in studies will also be simulated. |
| Benthic Metabolism Half-life or Anaerobic Aquatic Metabolism Half-life (days) and Reference Temperature | Diazinon | 73.5 at 20oC | MRID 46386602 | Representative half-life value from one study multiplied by standard three-fold factor to account for variability and uncertainty. |
| Aqueous Photolysis Half-life @ pH 7 (days) and Reference Latitude, 25oC | Diazinon | Stable (0) at 40oN (PRZM5/VVWM)  1e8 (PFAM) | MRID 48417202 | The aqueous photolysis half-life input value was adjusted for continuous illumination as well as for latitude/season to reflect photolysis in summer sunlight at 40o N latitude. |
| Hydrolysis Half-life (days) | Diazinon | Stable (0) (PRZM5/VVWM)  1e8 (PFAM) | MRID: 46235726 | Diazinon does undergo hydrolysis, see Section 3.A. The aquatic metabolism rates were not corrected for hydrolysis because the metabolism studies were conducted at different pH and temperatures than the hydrolysis studies. Therefore, diazinon was assumed to be stable to hydrolysis in modeling. Hydrolysis should be captured by the aerobic aquatic metabolism values because they were not corrected for hydrolysis. |
| Soil Half-life or Aerobic Soil Metabolism Half-life (days) and Reference Temperature | Diazinon | 34 at 25oC | MRIDs 46867004, 44746001, 46386605 | The 90 percent upper confidence bound on the mean of four half-life values. Values were adjusted to a temperature of 25oC using equation 1 in the PRZM-GW input parameter guidance (USEPA and Health Canada, 2013). The value of 34 days will be used for most of the modeling as there is the half-life was calculated for known diazinon residues. The 155 day value was based on half-lives estimated for diazinon plus lost radioactivity. The minimum value of 9 days and the maximum value of 155 days will be used to characterize uncertainty in this input parameter. |
| Diazinon plus lost residues | 155 at 25oC for characterization |  |
| Molecular Weight (g/mol) | Diazinon | 304.35 | -- | -- |
| Vapor Pressure (Torr) at 25oC | Diazinon | 7.22×10-5 | MRID 42970809, 40226101. | -- |
| Solubility in Water @ 25 OC, pH not reported (mg/L) | Diazinon | 65.5 | MRID 42970808 | -- |
| Foliar Half-life (days) | All | Stable (0) | Default | -- |
| Heat of Henry (Joules/mole) | Diazinon | 98,000 at 20oC | (Feigenbrugel *et al.*, 2004) | -- |
| Number of Applications | All | See input file | Diazinon Use Summary table | -- |
| Dates | All | Assumed based on type of application | Absolute and relative dates were used in modeling. |
| Amount |  | Diazinon Use Summary table | Maximum single application rate for the crop or use pattern |
| Application method |  | Diazinon Use Summary table | Incorporation depth was assumed to be the minimum incorporation depth for the use pattern. Runoff only occurs from the top 2 cm of soil (Carsel *et al.*, 1997). |
| Application Efficiency | All | Aerial: 0.95  Ground: 0.99 | Input parameter guidance (USEPA, 2009) | -- |
| Drift | All | See **Table 3.10** | Offsite transport guidance (USEPA, 2013) |  |
| PRZM5/VVWM Scenario | All | Determined based on Scenario-HUC matrix | -- | Screening scenario that is expected to result in a high end EEC. |

## Aquatic Modeling Results

The PRZM5/VVWM was used to determine diazinon concentrations in surface water and pore-water by simulating the maximum application rates and use patterns as described on diazinon labels. Each use pattern was simulated for HUC2 regions where the use occurs and aquatic bins 2 through 7. 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 3-1 Background Document: Aquatic Exposure Estimation**. Aquatic bin 1 represents aquatic habitats associated with terrestrial habitats and it was not simulated using aquatic modeling. Aquatic bin 9 is subtidal near shore habitat and aquatic bin 10 is the offshore marine habitat. EFED does not currently have models designed to estimate EECs for the estuarine/marine systems. EFED and the Services have assigned surrogate freshwater flowing or static systems to evaluate exposure in estuary and marine bins. Aquatic bin 5 will be used as a 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.

The estimated environmental concentrations (EECs) are based on labeled uses and associated application rates of diazinon. Average daily EECs were derived from the PRZM5/VVWM modeling for each HUC 2 and aquatic bin and are summarized in **Table 3-10** and **Table 3-11**. The range reflects peak EECs for maximum application rates and use patterns of diazinon as described on labels. The complete set of modeling results, including annual, overall, 4-day, 21-day, 60-day, and 90-day average water-column EECs and 21-day average pore-water EECs, are available in **APPENDIX 3-4b-d Aquatic Modeling EECs**. **APPENDIX 3-4e Aquatic Modeling EECs Figures with Effects Endpoints** provides an example of the temporal variability of EECs for a HUC 1 aquatic bin 7.

Table 3-10. Range of PRZM5/VVWM daily average water-column EECs

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **HUC 2** | **Range of 1-in-15 year Peak EECs (µg/L)** | | | | | |
| **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin7** |
| HUC 1 | 560 - 2,960 | 25.40 - 427 | 21.70 - 283 | 706 – 3,770 | 27.1 - 205 | 2.34 - 25.1 |
| HUC 2 | 559 – 2,290 | 62.10 - 366 | 52.70 - 329 | 704 – 2,950 | 27.0 - 142 | 2.81 - 19.2 |
| HUC 3 | 562 – 2,350 | 35.70 - 152 | 34.80 - 142 | 706 – 2,990 | 27.5 - 126 | 2.75 - 14.3 |
| HUC 4 | 561 – 2,340 | 25.10 - 350 | 23.00 - 346 | 706 – 2,910 | 27.3 - 141 | 2.88 - 21.3 |
| HUC 5 | 559 – 2,300 | 48.60 - 247 | 33.10 - 231 | 705 – 2,930 | 27.5 - 153 | 3.23 - 21.4 |
| HUC 6 | 562 – 2,330 | 35.30 - 223 | 34.20 - 218 | 707 – 2,960 | 27.1 - 122 | 2.39 - 11.3 |
| HUC 7 | 557 – 2,350 | 66.60 - 375 | 56.90 - 358 | 717 – 3,030 | 27.8 - 168 | 4.18 - 36.7 |
| HUC 8 | 555 – 2,290 | 44.40 - 386 | 43.20 - 375 | 697 – 2,890 | 26.6 - 116 | 2.47 - 14.2 |
| HUC 9 | 561 – 2,350 | 30.10 - 159 | 24.80 - 156 | 707 – 3,000 | 29.0 - 163 | 5.31 - 31.9 |
| HUC 10a | 561 – 2,320 | 33.50 - 128 | 28.90 - 105 | 712 – 2,950 | 58.4 - 219 | 23.8 - 86.6 |
| HUC 10b | 564 – 2,350 | 35.50 - 127 | 32.70 - 102 | 710 – 2,990 | 37.9 - 148 | 10.6 - 37.3 |
| HUC 11a | 557 – 2,410 | 63.40 - 306 | 70.90 - 318 | 709 – 3,080 | 83.3 - 510 | 33.3 - 229 |
| HUC 11b | 557 – 2,450 | 56.40 - 340 | 59.90 - 346 | 701 – 3,090 | 55.4 - 445 | 10.9 - 201 |
| HUC 12a | 556 – 2,380 | 41.80 - 315 | 40.40 - 327 | 701 – 3,030 | 39.8 - 334 | 10.8 - 136 |
| HUC 12b | 554 – 2,400 | 41.90 - 330 | 39.50 - 328 | 695 – 3,030 | 36.2 - 217 | 8.94 - 84.8 |
| HUC 13 | 551 – 2,410 | 40.10 - 377 | 39.80 - 323 | 871 – 5,310 | 56.1 – 1,230 | 9.26 - 478 |
| HUC 14 | 566 – 2,390 | 65.60 - 288 | 59.50 - 284 | 716 – 3,050 | 65.1 - 721 | 28.9 - 324 |
| HUC 15a | 557 – 2,350 | 38.10 - 443 | 35.20 - 435 | 734 – 6,070 | 218 – 2,810 | 45.5 - 592 |
| HUC 15b | 544 – 2,290 | 38.10 - 422 | 35.00 - 406 | 716 – 3,370 | 84.6 - 1040 | 20.8 - 231 |
| HUC 16a | 564 – 2,360 | 70.90 - 302 | 31.70 - 174 | 718 – 2,980 | 46.8 - 350 | 17.9 - 141 |
| HUC 16b | 564 – 2,370 | 61.10 - 283 | 32.10 - 185 | 710 – 3,000 | 52.2 - 409 | 10.7 - 189 |
| HUC 17a | 306 – 2,320 | 22.20 - 191 | 20.70 - 170 | 386 – 2,960 | 59.2 - 636 | 26.2 - 351 |
| HUC 17b | 308 – 2,340 | 15.20 - 215 | 14.10 - 202 | 387 – 2,960 | 15.0 - 298 | 2.99 - 94.4 |
| HUC 18a | 306 – 2,370 | 41.40 - 918 | 40.80 - 947 | 401 – 3,090 | 54.7 - 341 | 23.7 - 159 |
| HUC 18b | 305 – 2,350 | 38.50 - 720 | 38.90 - 776 | 386 – 2,980 | 27.1 - 301 | 10.5 - 121 |
| HUC 19a | 574 – 2,420 | 37.40 - 165 | 34.50 - 143 | 729 – 3,090 | 29.8 - 137 | 6.12 - 32.4 |
| HUC 19b | 573 – 2,350 | 41.60 - 188 | 33.20 - 162 | 732 – 3,070 | 33.7 - 177 | 9.17 - 56.7 |
| HUC 20a | 564 – 2,270 | 44.80 - 242 | 44.90 - 249 | 777 – 3,120 | 75.5 - 307 | 31.7 - 122 |
| HUC 20b | 562 – 2,280 | 33.60 - 193 | 36.00 - 201 | 712 – 2,860 | 46.4 - 150 | 14.3 - 52.3 |
| HUC 21 | 560 – 2,260 | 39.90 - 272 | 39.50 - 260 | 703 – 2,860 | 26.9 - 112 | 2.27 - 13 |
| 1 The lowest and highest peak EECs of all simulated use patterns is reported for each HUC2 and aquatic bin. | | | | | | |

Table 3-11. Range of PRZM5/VVWM pore-water EECs

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **HUC 2** | **Range of 1-in-15 year Peak EECs (µg/L)** | | | | | |
| **Bin 2** | **Bin 3** | **Bin 4** | **Bin 5** | **Bin 6** | **Bin7** |
| HUC 1 | 27.3 - 596 | 6.27 - 3,010 | 6.39 - 3,940 | 36.6 - 328 | 5.32 - 48.2 | 0.80 - 9.08 |
| HUC 2 | 25.3 - 306 | 11.60 - 843 | 9.26 - 820 | 35.8 - 307 | 4.97 - 45.6 | 0.66 - 6.42 |
| HUC 3 | 27.2 - 170 | 5.72 - 191 | 6.00 - 150 | 35.4 - 248 | 5.87 - 35.8 | 0.95 - 4.76 |
| HUC 4 | 27.8 - 233 | 3.32 - 444 | 2.72 - 445 | 36.9 - 316 | 4.72 - 42.8 | 0.79 - 6.14 |
| HUC 5 | 28.2 - 218 | 15.10 - 216 | 14.10 - 161 | 35.2 - 303 | 5.14 - 45.4 | 1.00 - 7.12 |
| HUC 6 | 27.5 - 145 | 7.97 - 117 | 10.00 - 119 | 36.0 - 225 | 5.63 - 29.8 | 0.70 - 3.9 |
| HUC 7 | 26.2 - 217 | 9.16 - 1,010 | 10.10 - 1,250 | 35.5 - 288 | 5.09 - 45.7 | 0.95 - 12.1 |
| HUC 8 | 22.3 - 112 | 9.41 - 75 | 7.11 - 56 | 29.9 - 149 | 3.26 - 20.1 | 0.35 - 2.87 |
| HUC 9 | 27.3 - 235 | 28.00 - 897 | 22.60 - 940 | 38.6 - 327 | 5.98 - 47.7 | 1.45 - 9.73 |
| HUC 10a | 26.2 - 166 | 12.20 - 334 | 11.30 - 249 | 36.5 - 207 | 13.2 – 61.0 | 6.13 - 27.3 |
| HUC 10b | 28.4 - 172 | 8.02 - 122 | 9.35 - 91 | 39.8 - 237 | 10.8 - 47.3 | 4.00 - 14.8 |
| HUC 11a | 26.3 - 239 | 16.90 - 177 | 23.70 - 320 | 33.9 - 348 | 15.1 - 150 | 7.47 - 76.8 |
| HUC 11b | 25.3 - 286 | 10.20 - 166 | 16.30 - 240 | 33.6 - 385 | 11.3 - 115 | 4.37 - 50.1 |
| HUC 12a | 25 - 224 | 8.65 - 233 | 8.95 - 341 | 32.1 - 313 | 6.61 - 87.6 | 2.17 - 35.4 |
| HUC 12b | 22.2 - 233 | 7.80 - 161 | 9.93 - 249 | 30.2 - 318 | 7.48 - 60.6 | 3.02 - 21.3 |
| HUC 13 | 21.6 - 248 | 8.71 - 266 | 4.53 - 45 | 50.2 - 382 | 15.5 - 204 | 2.99 - 93.3 |
| HUC 14 | 28.8 - 243 | 11.10 - 141 | 9.94 - 84 | 51.1 - 359 | 22.6 - 160 | 10.4 - 79.3 |
| HUC 15a | 22.6 - 289 | 8.29 - 569 | 5.85 - 403 | 57.7 - 467 | 40.7 - 659 | 9.74 - 195 |
| HUC 15b | 26.3 - 232 | 6.86 - 374 | 5.79 - 275 | 45.6 - 210 | 23.8 - 168 | 4.96 - 43.4 |
| HUC 16a | 28.7 - 177 | 19.30 - 86 | 7.05 - 28 | 38.7 - 267 | 12.3 - 83.3 | 6.15 - 39.8 |
| HUC 16b | 29.6 - 191 | 12.30 - 125 | 6.69 - 27 | 39.6 - 351 | 18.0 - 107 | 3.65 - 50.2 |
| HUC 17a | 15.3 - 220 | 10.90 - 274 | 13.40 - 243 | 20.4 - 319 | 20.6 - 184 | 9.54 - 118 |
| HUC 17b | 16.8 - 252 | 2.65 - 120 | 3.60 - 136 | 21.3 - 340 | 3.76 - 89.4 | 0.87 - 33.4 |
| HUC 18a | 15.6 - 202 | 11.90 - 378 | 14.30 - 887 | 23.7 - 273 | 13.3 - 97.8 | 7.17 - 49.3 |
| HUC 18b | 15 - 214 | 9.27 - 330 | 10.50 - 763 | 20.0 - 270 | 7.15 - 60.9 | 2.96 - 29.1 |
| HUC 19a | 30.9 - 230 | 3.96 - 75 | 4.66 - 78 | 48.5 - 344 | 6.94 - 49.2 | 1.65 - 10.6 |
| HUC 19b | 30.9 - 168 | 5.31 - 99 | 5.76 - 103 | 47.1 - 325 | 9.05 - 77.5 | 2.79 - 34.5 |
| HUC 20a | 24.4 - 110 | 9.15 - 42 | 10.80 - 45 | 40.8 - 174 | 13.4 - 50.3 | 6.36 - 22.8 |
| HUC 20b | 23.5 - 115 | 3.83 - 27 | 4.42 - 33 | 33.2 - 147 | 10.2 - 26.3 | 2.61 - 11.2 |
| HUC 21 | 22.7 - 103 | 3.21 - 188 | 3.23 - 204 | 29.1 - 137 | 3.17 - 14.7 | 0.41 - 2.2 |
| 1 The lowest and highest peak EECs of all simulated use patterns is reported for each HUC2 and aquatic bin. | | | | | | |

In the draft BEs, there was little confidence in the EECs derived using the PRZM5/VVWM model for Bin 3 (moderate flow aquatic bin) and Bin 4 (high flow aquatic bin) as (a) the maximum EECs exceeded the active ingredient’s water solubility limit, (b) the EECs were higher than those estimated for Bin 2, which should not have occurred as the higher flowrates in Bins 3 and 4 should have contributed to dilution as well as advective dispersion, (c) the EECs for Bin 3 were higher than those estimated for Bin 4, which again, given the higher flowrate for Bin 4, was contrary to what one would expect, and (d) the EECs were higher, by several orders of magnitude, than those derived for the static bins, which have no outlet for the release of pesticide. As a result, a qualitative approach was considered in the draft BEs, where Bin 2 EECs were generated using the PRZM5/VVWM, Bin3 EECs were characterized as being conservatively 5 and 10 times lower than the Bin 2 EECs, and the Bin 4 EECs were characterized as being conservatively 5 and 10 times lower than the Bin 3 EECs.

During the public comment period, recommendations were discussed that were subsequently incorporated into EFED’s methodology for estimating PRZM5/VVWM model generated Bin 3 and 4 EECs.

Perhaps most importantly, daily (24-hour) mean concentrations have been adopted in place of the initial (time zero) concentrations that EFED had previously employed as acute EECs. 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 EFED’s Index Reservoir, or is assumed to have no flow out of it, which is the case for the EFED 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. 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.

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 bin 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 EFED is considering this as a potential, future modification of the new methodology. 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, EFED has not incorporated this refinement into the BE’s for the three OP’s; however, EFED does intend to incorporate this adjustment into the BEs for carbaryl and methomyl.

Simulations using different meteorological data for different wet-harvested cranberry growing areas results in similar EECs. Results from PFAM indicate peak 1-in-15 year aquatic EECs are 61.9 to 171 µg/L for concentrations of diazinon in cranberry bogs and are generally in the range of EECs generated using PRZM5/VVWM for the different HUC2 regions and static aquatic bins (0.351 to 424 µg/L) **Table 3-12**. Peak EECs occurred during the winter flood in January, and not during the three day harvest flood simulated in October. Peak aquatic EECs in the harvest flood water ranged from 3.01 to 61.4 µg/L. These simulated concentrations are approximately half those observed in an irrigation ditch adjacent to a cranberry bog treated with diazinon, where the maximum concentration detected was 456 µg/L; however, the maximum application rate used in the study 5.35 lbs a.i./A applied twice and the maximum application rate simulated was 3 lbs a.i./A applied three times with a retreatment interval of 14 days. Additionally, the maximum monitored concentration was detected just after the second application and the application was made to a flooded bog. In the PFAM simulation, the last application occurred in August and the flood did not occur until harvest in October, as this is the typical flood pattern for cranberry bogs.

Table 3-12. Summary of 1-in-15 year peak EEC in cranberry bogs

| State of Simulation | 1-in-15 year Peak EEC in µg/L | |
| --- | --- | --- |
| Winter Flood | Harvest Flood |
| Connecticut | 140 | 44.3 |
| New Jersey | 103 | 36.2 |
| Wisconsin A | 171 | 48.6 |
| Wisconsin B | 61.9 | 3.01 |
| Oregon | 132 | 61.4 |

## Aquatic Modeling Sensitivity Analysis

A key recommendation of the NAS report on ESA was to characterize model sensitivities and to quantify, where possible, the impact of the assumptions surrounding those inputs on model outputs. In the case of EPA’s aquatic exposure assessment, the model sensitivities have been examined and documented by various agencies (Carbone *et al.*, 2002; USEPA, 2004; Young, 2014). The sensitivity of the various input parameters was also evaluated during the development of the underlying models PRZM5 and the VVWM (replacement model for EXAMS) (Burns, 2004; FEMVTF, 2001).

Pesticide runoff is sensitive to a combination of factors including pesticide application date, application method, curve number, pesticide degradation rate, pesticide sorption coefficient, and rainfall timing and amount. In more arid regions such as California, spray drift may contribute more to surface water EECs than runoff. The California Department of Pesticide Regulation’s evaluation of pesticide runoff has indicated that as much as 95% of the variability in surface runoff from PRZM5 can be accounted for by a few select fate parameters and the curve number (Luo and Deng, 2012). Curve number is an empirical parameter used in PRZM5 to predict direct runoff from a field. A curve number is dependent on the hydrologic soil group, land use treatment, or cover, and the hydrologic condition of the field (*i.e.*, poor or good). Pesticide runoff is also sensitive to the application rate, as the potential for runoff and drift increases with greater chemical application. However, as EPA regulates pesticides based on the label to ensure that the maximum label rate does not result in adverse effects to aquatic species, the application rate is not considered a factor in the sensitivity of the modeling but rather a factor examined during risk mitigation.

For waterbodies, the input parameters having the greatest effect on the EEC are aerobic aquatic metabolism, sorption, and to a lesser extent aqueous photolysis and volatility. Photolysis is of lesser importance because it is sensitive to light intensity, and thus more active in the upper portion of the water column. While EECs in a waterbody are expected to be lower for volatile chemicals, detections of volatile chemicals would only likely be observed if the chemical is transported to the waterbody. In flowing waterbodies, flow rate through the waterbody also has an impact on EECs, as the lower the flow rate, the longer the residence time in the waterbody and the higher the concentration.

For surface water modeling, the variability and uncertainty in the model input parameters and their impact on EECs are captured in a number of ways. Variability in model output due to curve number selection is captured by the spatial variation in soil types represented by EPA’s PRZM5/VVWM scenarios. The sensitivity to rainfall timing and intensity is similarly captured by varying scenarios across the landscape and also by utilizing multiple years of meteorological data (*i.e.*, precipitation). Sensitivity to application date selection is captured by varying the selected application date across a window of anticipated application dates typically derived from a variety of sources including label information, USDA Crop Profiles, USDA Usual Planting and Harvesting Dates, Usage data (both public and proprietary), and available information on pest pressures, such as information available in the crop profiles from the Integrated Pest Management Center ([http://www.ipmcenters.org//index.cfm/center-products/crop-profiles/](http://www.ipmcenters.org/index.cfm/center-products/crop-profiles/)). Finally, EPA typically has the ability to characterize the potential influence of known variability in key fate input parameters, and explore alternative assumptions. For example, **Table 3-13** shows the range of soil metabolism and sorption properties derived for diazinon that can be varied for sensitivity analysis purposes.

Table 3-13. Parameter sensitivity analysis for diazinon

|  |  |  |  |
| --- | --- | --- | --- |
| **Parameter** | **Modeled1** | **High KOC, Low Persistence** | **Low KOC, High Persistence** |
| KOC (mL/g o.c.) | 824 | 3779 | 138 |
| Aerobic aquatic t1/2 (days) | 13.2 | 10 | 16.3 |
| Anaerobic aquatic t1/2 (days)3 | 73.5 | 24.5 | 73.5 |
| Aerobic soil t1/2 (days)2 | 34 | 9 | 387\* |

1. Input parameters used in developing reported EECs (**Table 3-9**). Low are the minimum values reported in fate table, while high are the maximum values.
2. The high value accounts for parent and lost radioactivity. MRID 46867004 was not considered appropriate for modeling because only one replicate was available in the study.
3. Only one anaerobic aquatic metabolism value is available. Based on input parameter guidance (USEPA, 2009a) the value was multiple by three in the main simulation.

EPA currently employs an approach that selects scenarios, application timing and chemical properties from a distribution of available data that are intended to provide reasonable upper bound estimates of exposure. In order to address the NAS recommendations, EPA evaluated the impact of the current assumptions within the range of available data. EPA employed this type of analysis for representative scenarios within the Biological Evaluation (BE) to provide a sense of how the EECs can vary based on these parameters. The variables selected capture the impact of alternate assumptions of vulnerability using varying assumptions of application timing and fate inputs. The model input parameters selected for the parameter sensitivity analysis include those summarized in **Table 3-13** as well as application timing (see discussion below). The sensitivity analysis provides information on how much higher or lower the EECs could be with alternative assumptions.

EPA believes that the estimated modeled exposures provide reasonable high-end estimates of exposure and represent a good predictor of upper level pesticide concentrations. Typically, the Agency evaluates exposure using the EEC based on the 1-in-10 year return frequency. The use of the overall maximum EEC from a 30-year simulation run, while protective, represents a peak value that occurs rather infrequently (*i.e.*, one day in 30 years). In the case of the pilot chemicals, the 1-in-15 year return frequency has been selected to reflect the need to characterize the likelihood of an adverse effect during the course of the federal action, which has a defined duration of 15 years based on the registration review cycle.

The PRZM5/VVWM scenarios used in the modeling have been developed to represent a combination of factors that can be reasonably expected to occur, although the combination of these parameters is expected to result in EECs in the upper end of the distribution. For example, the PRZM5/VVWM scenarios are developed to represent a combination of soil hydrologic group and land cover type to yield a high end curve number for a use site, which would result in maximum plausible runoff and erosion from the area. This combination is expected to occur within a given area; however, it is feasible that other combinations of soil and land cover types that are characteristic of a lower curve number may occur in other areas. Variation in curve number is captured by using a large suite of PRZM5/VVWM scenarios to represent variability across the landscape. More details on scenarios and scenario development may be found at: [http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment#aquatic](http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/models-pesticide-risk-assessment%23aquatic).

Similarly, EPA selects chemical-specific model inputs in an effort to ensure exposure is not underestimated by selecting a chemical input value from somewhere in the upper, rather than lower, tail of possible mean half-lives. As a result, most characterization of model uncertainty for these parameters tends to be on the less conservative side. Details on model inputs can be found at: <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling>

**Table 3-4** summarizes the sensitivity analysis results when varying fate parameters as described in **Table 3-13** and by simulating a ten and 100-foot spray drift buffer for an aerial and ground application. The top row (main simulation) shows the EECs for the ornamental scenario assuming one application and the standard fate parameters (‘modeled’ values in **Table 3-13**) and application date assumed in the main simulation[[16]](#footnote-17) simulated. The sensitivity simulation EECs may be compared with the main simulation EECs to see whether varying the fate parameter inputs results in higher or lower EECs. This is explored for one application per year and use scenario across all HUCs and bins.

Overall varying the metabolism inputs resulted in EECs within 20% of the main simulation (**Figure 3-2** and **Table 3-14**). High and low aerobic aquatic metabolism inputs resulted in EECs within 10% (usually within 1%) of the main simulation. High and low aerobic soil metabolism inputs resulted in EECs within 20% of the main simulation. Assuming a low anaerobic aquatic input resulted in EECs within 1% of the main simulation.

Varying the KOCs had more of an impact on the EECs. The low KOC (138 L/kg-oc) simulation EECs and low KOC with high persistence simulation EECs are up to 2.7 times the main simulation EECs. By contrast, use of the highest KOC of 3,779 L/kg-oc in the simulation resulted in peak EECs as low as 0.25 times the main simulation peak EEC.

The maximum peak EECs for each aquatic bin, when simulating a 10 foot or 100 foot buffer and no spray drift, ranged from 0.69 to 1.02[[17]](#footnote-18) the main simulation peak EEC, with the impact of the buffer having the most reduction in EEC for bin 5. These buffers were on some labels as recommendations to applicators to protect sensitive aquatic areas.

The fate sensitivity analyses show that changing the fate inputs in the range of possible inputs results in both lower and higher EECs as compared to the main simulation. The most sensitive fate input parameter was the sorption coefficient, followed by aerobic soil and anaerobic aquatic metabolism input values. The EECs would likely be more sensitive to the aerobic aquatic metabolism input value if a wider range was observed in the measured aerobic aquatic input values (measured values ranged from 10 to 16.3 days).

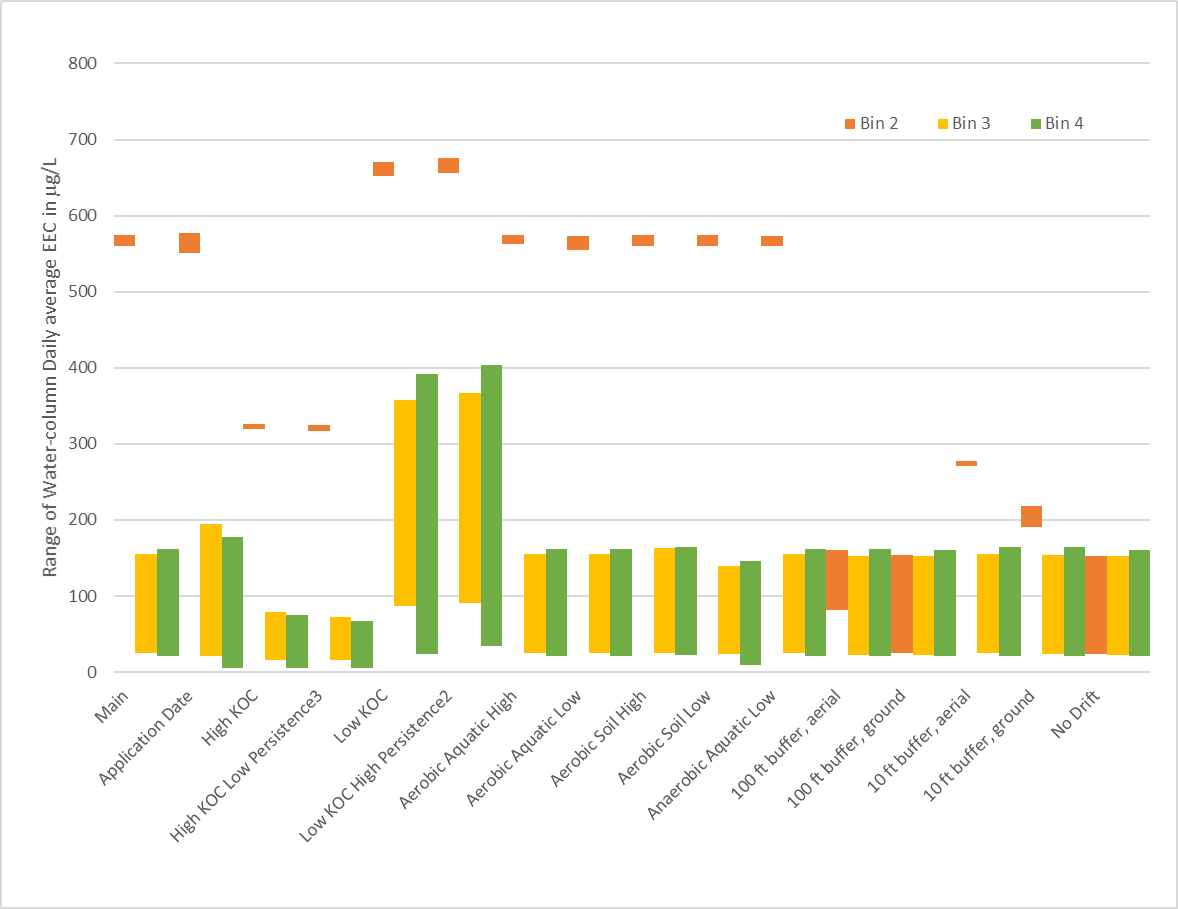
Table 3-14. Sensitivity of EECs to application date and fate parameters1

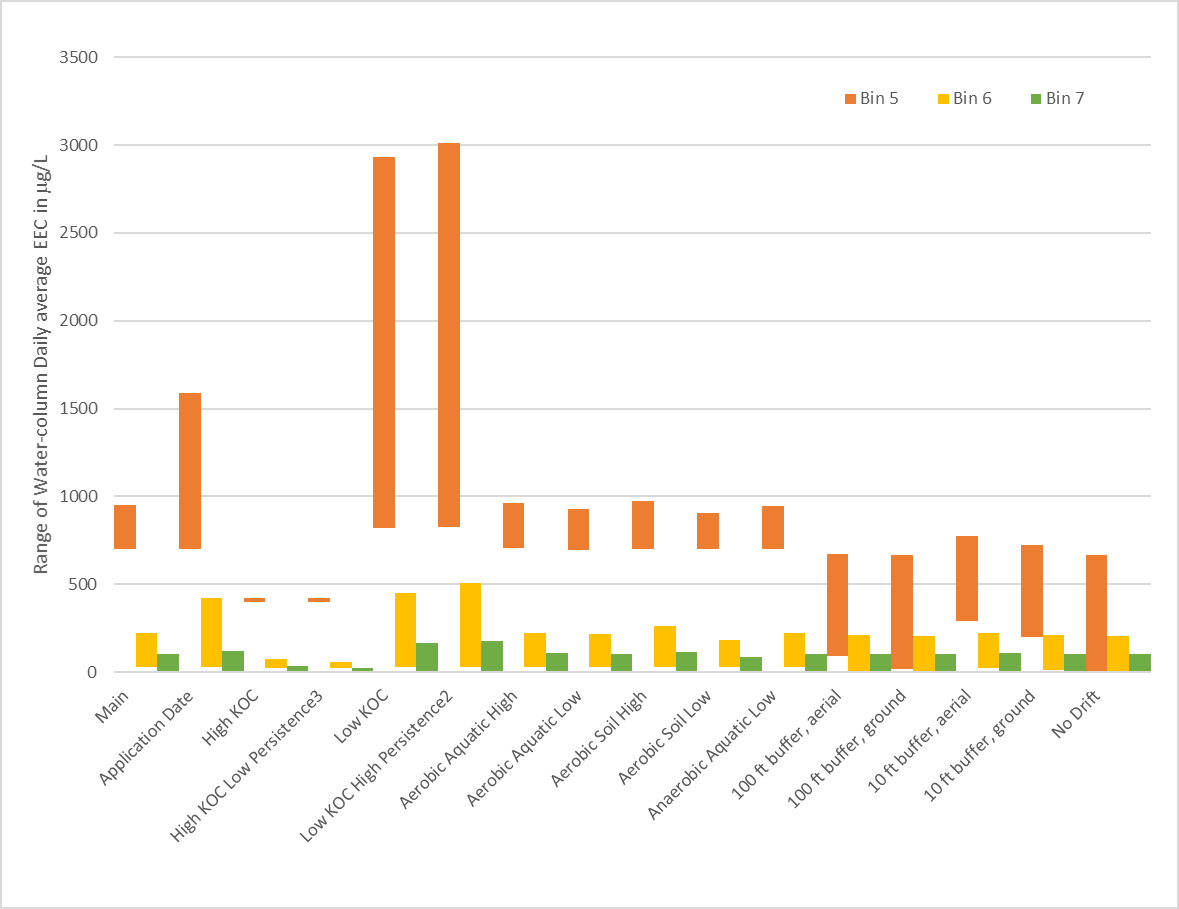
|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Range of Daily Average (1-in-15 year) EEC (µg/L) for Each Aquatic Bin | | | | | |
| Parameter Simulated | 2 | 3 | 4 | 5 | 6 | 7 |
| Main | 560 - 574 | 25.1 – 155 | 21.7 - 162 | 703 – 949 | 26.9 – 223 | 2.27 – 105 |
| Application Date | 551 – 577 | 21.2 – 194 | 4.85 - 177 | 698 – 1,590 | 26.8 – 419 | 2.26 – 118 |
| High KOC | 320 – 326 | 15.3 – 78.9 | 5.57 – 74.5 | 401 – 423 | 25 – 72.1 | 2.17 – 36 |
| High KOC Low Persistence3 | 317 – 325 | 15.3 – 72.5 | 4.87 – 66.6 | 399 – 420 | 24.8 – 58.4 | 2.15 – 25.6 |
| Low KOC | 652 – 670 | 86.5 – 358 | 23.2 - 392 | 819 – 2,930 | 27.4 – 452 | 3.02 – 166 |
| Low KOC High Persistence2 | 656 – 676 | 90.1 – 367 | 33.9 - 403 | 824 – 3,010 | 27.5 – 508 | 3.05 – 175 |
| Aerobic Aquatic High | 563 – 575 | 25.2 – 155 | 21.7 - 162 | 707 – 961 | 27 – 225 | 2.35 – 106 |
| Aerobic Aquatic Low | 555 – 573 | 25.1 – 155 | 21.7 - 162 | 697 – 930 | 26.6 – 219 | 2.24 – 104 |
| Aerobic Soil High | 560 – 574 | 25.4 – 163 | 22.9 - 165 | 703 – 975 | 26.9 – 262 | 2.33 – 113 |
| Aerobic Soil Low | 560 – 574 | 24.3 – 139 | 9.62 - 146 | 703 – 907 | 26.9 – 182 | 2.26 – 84.3 |
| Anaerobic Aquatic Low | 560 – 573 | 25.1 – 155 | 21.7 - 162 | 703 – 948 | 26.9 – 220 | 2.27 – 104 |
| 100 ft buffer, aerial | 81.3 – 161 | 21.8 – 153 | 21.7 - 162 | 90.4 – 673 | 8.53 – 210 | 2.17 – 105 |
| 100 ft buffer, ground | 25.3 – 154 | 21.8 – 153 | 21.7 – 161 | 18.1 – 669 | 2.32 – 207 | 0.78 – 104 |
| 10 ft buffer, aerial | 271 – 278 | 25.5 – 155 | 21.7 – 165 | 289 – 777 | 24.5 – 221 | 4.31 – 107 |
| 10 ft buffer, ground | 190 – 218 | 23.5 – 154 | 21.7 – 164 | 199 – 722 | 12.8 – 212 | 1.71 – 105 |
| No Drift | 23.1 – 152 | 21.8 – 152 | 21.7 - 161 | 0.86 – 669 | 0.79 – 206 | 0.42 - 103 |

1 Applications to ornamentals grown in nurseries with a diazinon application of 1 lbs a.i./A applied one time a year over 30 years by ground was simulated with differences in the fate input assumptions (as described in **Table 3‑13**) and by simulating a 10-foot buffer or 100-foot buffer with an aerial or ground application assumption, and simulating the main simulation with and without drift. In the main simulation, results reflect the inputs simulated for estimating the aquatic EECs used in the risk characterization and assuming a ground application. The simulations were run for all HUC 2 regions and the range for each aquatic bin shows the minimum and maximum peak EEC across HUCs for each bin.

2 Fate parameters of KOC=138 L/kg-oc, aerobic aquatic metabolism T1/2=16.3 d, anaerobic aquatic metabolism t1/2=73.5 d, and the aerobic soil metabolism t1/2=387 d.

3 Fate parameters of KOC=3779 L/kg-oc, aerobic aquatic metabolism T1/2=10, anaerobic aquatic metabolism t1/2=24.5 d, and the aerobic soil metabolism t1/2=9 d.



******Figure 3-2. Range of the daily average 1-in-15 year EECs for different aquatic bins simulated in the fate inputs sensitivity analysis.**

EPA evaluated the sensitivity of the application date by varying it across a 365-day window of time for the ornamental simulation using the input assumptions for the main model runs. This was completed for all HUC2 and aquatic bin combinations. The ornamental scenario was chosen because it involves 1 application per year, so it may be simulated for all 365 days and it was simulated in all HUC 2 regions[[18]](#footnote-19). A summary of the variability in outputs for a representative scenario is captured in **Table 3-14**. Complete results are provided in **APPENDIX 3-4d Aquatic EECs Sensitivity Analysis**. Results suggest that an alternative application date could increase peak EECs by up to up to 1.88x. These results illustrate the importance of the application date simulated. An interesting result is that the month that resulted in the highest EEC varied between the aquatic bins (**Table 3-15**). **Figure 3-3** summarizes the maximum peak EEC across HUC 2 regions for aquatic bins 2, 5, 6, and 7. As expected with different meteorological stations used to simulate different HUC 2 regions, the application month resulting in the highest EEC also varied between different HUC 2 regions (**Figure 3-4**).

Table 3-15. Summary of highest peak EEC resulting from different application dates chosen for different aquatic bins

| **Month** | **Daily Average (1-in-15 year) EEC (µg/L) for Each Aquatic Bin1** | | | | | | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Bin 2** | **Bin 3** | | **Bin 4** | **Bin 5** | **Bin 6** | **Bin 7** |
| January | 577 | | 187 | 175 | 950 | 357 | 95 |
| February | 576 | | 194 | 177 | 913 | 419 | 91.1 |
| March | 576 | | 176 | 151 | 985 | 201 | 61.7 |
| April | 575 | | 137 | 131 | 935 | 191 | 74.3 |
| May | 574 | | 138 | 77 | 857 | 147 | 65.9 |
| June | 573 | | 139 | 78 | 988 | 207 | 67.8 |
| July | 571 | | 130 | 78 | 1350 | 337 | 69.2 |
| August | 570 | | 135 | 81 | 1180 | 237 | 70.1 |
| September | 572 | | 136 | 108 | 1590 | 321 | 99 |
| October | 574 | | 144 | 151 | 931 | 323 | 66.9 |
| November | 576 | | 185 | 172 | 1160 | 348 | 97 |
| December | 576 | | 192 | 172 | 947 | 368 | 118 |

\*Values in red and *italics* are the highest value for that aquatic bin.

1 The highest peak EECs that occurred for simulations with an application date in the month specified is provided for each aquatic bin.

Figure 3-3. Application date sensitivity analysis across aquatic bins (ornamental scenario across all HUC2s)

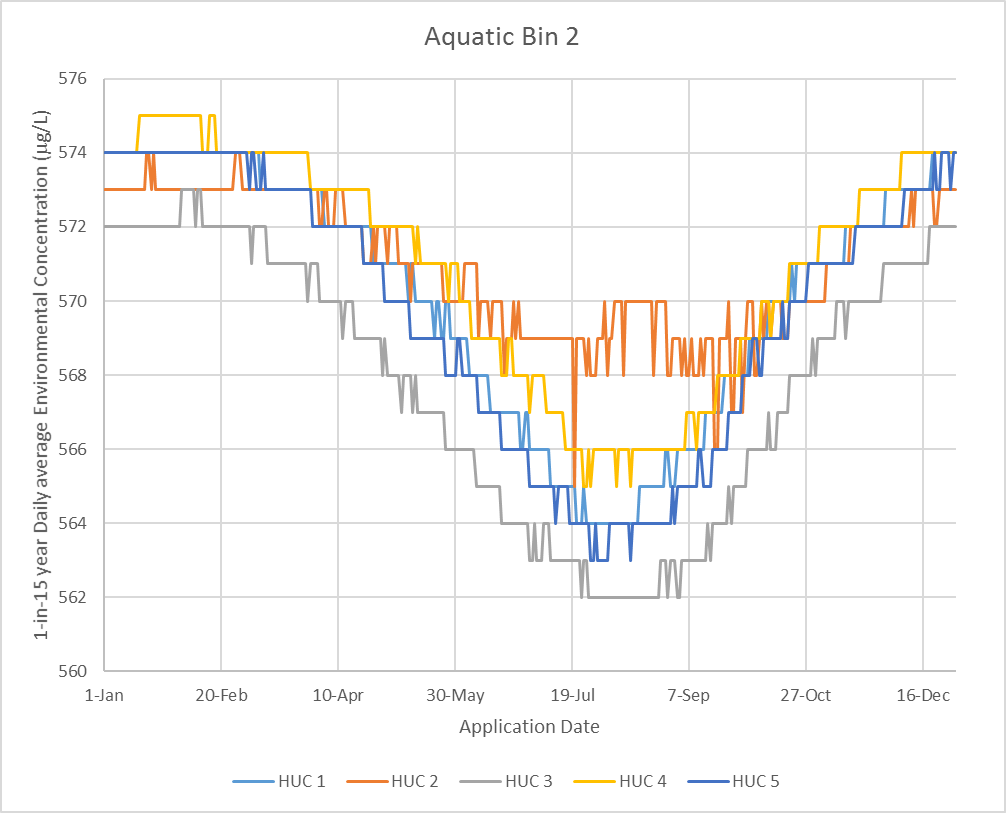
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Figure 3-4. Application date sensitivity analysis across selected HUC2 regions (ornamental scenario bin 2)

## Available Monitoring Data

General monitoring concentrations in waterbodies in the United States, irrespective of the size or location of the waterbody, ranged from not detected to 61.9 µg/L (**Table 3-18**). Data from these monitoring studies are not correlated with known applications of pesticides under well-described conditions (*e.g.*, application rate, field characteristics, water characteristics, and meteorological conditions). Therefore, general monitoring data cannot be used to estimate pesticide concentrations after a pesticide application or to evaluate performance of fate and transport models (NRC, 2013). While general monitoring data may underestimate potential exposure, they provide useful information for describing water quality trends and the environmental baseline condition of species habitats including the occurrence of chemical mixtures and the presence of abiotic stressors that can increase risk.

### Field Studies

There were two field scale monitoring studies where the application were known and residues in water following applications were followed. These are summarized briefly in the monitoring section and discussed in more detail in **APPENDIX 3-1. Environmental Transport and Fate Characterization**. Three field dissipation studies were submitted where diazinon was applied to three apple orchards in Pennsylvania and diazinon residues were monitored in adjacent ponds (MRID 41490401, 41490402, 41490403). Maximum concentrations measured in ponds ranged from 12.8 to 113.0 µg/L in water and there were detectable residues in sediment at two sites. The ponds were similar in size to aquatic bins 6 and 7 where modeled EECs ranged from 0.75 to 2,910 µg/L. The application rate at the orchards was 3 lbs a.i./A, which is higher than the current recommended rate of 2 lbs a.i./A. Szeto *et al.* (1990) evaluated concentrations of diazinon and diazoxon in cranberry bogs and adjacent waters after application of Diazinon 5G (a granular formulation)[[19]](#footnote-20). The maximum diazinon concentration in water detected was 456 µg/L in irrigation ditches which decreased to below 100 µg/L within three to four days after treatment. Concentrations in the adjacent reservoir were lower with a maximum of 78.5 µg/L. Szeto *et al.* (1990) indicated residues observed in tributaries were much lower and were likely caused by leakage from the irrigation water through the gate between the reservoir and the waterways.

### General Monitoring Data

There are several monitoring studies and data from several sources available on diazinon residues in drinking water (raw and finished), surface water, groundwater, sediment, tissue (fish and mussels), air, rain, and snow. Although samples were collected in agricultural areas during the season of pesticide use, most studies were not specifically targeted at diazinon and the frequency of sample collection was not adequate to ensure the capture of peak concentrations. The data are useful in that they provide some information on the occurrence of diazinon in the environment under existing usage conditions. However, the measured concentrations should not be interpreted as reflecting the upper end of potential exposures. Field study monitoring, wherein application dates and amounts of applied materials are known and concentrations are followed in relation to the application(s), are discussed in the summarized field dissipation data.

Changes in diazinon use patterns were implemented between 2004 and 2008, after the Reregistration Eligibility Decision was completed (see **Section 1.1.2.5 Outstanding Mitigations of the Problem Formulation**). Updates included cancellation of non-agricultural uses (except nurseries), seed treatment uses, granular formulations, and only allowing aerial applications of diazinon to lettuce.[[20]](#footnote-21) Thus, monitoring conducted prior to this period may not reflect current use patterns of diazinon. In order to evaluate whether changes in the observed monitoring results reflect changes in use patterns, the frequency, location, and timing of monitoring and how it relates to usage information in the area monitored must be considered.

Diazinon is identified as a cause of impairment for 59 water bodies listed as impaired under section 303(d) of the Clean Water Act in California, Kansas, Oklahoma, and Washington.[[21]](#footnote-22) Impaired waters include rivers, creeks, drains, sloughs, channels, lakes, harbors, and drainage ditches. There are 107 Total Maximum Daily Loads (TMDL) listed for diazinon in California. Section 304(a) ambient water quality criteria[[22]](#footnote-23), aquatic life benchmarks, and Health Advisory levels[[23]](#footnote-24) (**Table 3-17**), have been established for diazinon. Monitoring data, impaired waters, and TMDLs for diazinon demonstrate that the use of diazinon may result in transport of diazinon to surface water at levels that may cause risk to human health and/or ecological receptors.

Table 3-17. Office of Water health advisories for diazinon1

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Health Advisories2 | | | | | |
| 10-kg Child | | 70-kg Adult | | | |
| 1-day (µg/L) | 10-day  (µg/L) | RfD  (mg/kg/day) | DWEL  (µg/L) | Life-time (µg/L) | mg/L at 10-4 Cancer Risk |
| 20 | 20 | 0.0002 | 7 | 1 | NA |

DWEL=Drinking Water Equivalent Level

RfD=Reference Dose  
1 The 2012 Edition of the Drinking Water Standards and Health Advisories is available at: <http://water.epa.gov/action/advisories/drinking/upload/dwstandards2012.pdf> (accessed 2/28/2015)

2 Health advisories, sponsored by the EPA’s Office of Water (OW), are concentrations of drinking water contaminants at which adverse health effects are not anticipated to occur over specified exposure durations.

#### Surface Water

Diazinon is one of the most frequently detected pesticides in surface water, and has been detected in 46 states (**Figure 3-7**), in every major U.S. river basin (including the Mississippi, Columbia, Rio Grande, and Colorado rivers), and in miscellaneous waters including various large rivers and major aquifers. The highest diazinon concentration reported was 61.9 µg/L, detected in a creek in California in 2009 (**Table 3-18**). Concentrations detected in tributaries to rivers are generally higher than those in rivers (Munoz *et al.*, 2011; Starner, 2009). Eleven states[[24]](#footnote-25) had surface water detections at 0.9 µg/L or greater, and 34 states had detections above 0.1 µg/L. Generally, the greater the number of samples collected, the higher the concentrations detected within a given state. Detections above 1 µg/L have continued to occur since 2007, when several mitigations on diazinon use[[25]](#footnote-26) were implemented; concentrations above 0.1 µg/L are common, especially in high use areas (**Figure 3-5**). Higher diazinon concentrations and higher frequencies of detection are generally observed in high use areas following precipitation events. For example, in California higher concentrations and detection frequencies (up to 90% detection frequency) were found in the Salinas and Imperial Valleys where lettuce, which receives the highest pounds of diazinon annually (in the U.S. between 2004 and 2012), is grown (Starner, 2009). In California, diazinon has been detected in areas with high and moderate irrigation season agricultural use, and in areas where orchards are grown and diazinon is commonly applied in the dormant season (November through February). Crops commonly grown in these areas include lettuce, spinach, broccoli, and other cool season crops (Starner, 2009), as well as almonds and stone fruit. There are two datasets that were specific to drinking water, and these are discussed in more detail below. Geospatial analysis suggests that some detections may also have occurred near the locations of drinking water treatment plant intakes. The number of samples collected per year across the United States has also varied over time, with a reduction in the number of samples collected in recent years, especially in the NAWQA data set (**Figure 3-6**).

Diazoxon was also detected in surface water, at a maximum concentration of 0.43 µg/L. The detection frequency of diazoxon in surface water is lower (0.2 to 6%) than that for parent diazinon. In surface water monitoring data wherein residues of both diazinon and diazoxon were detected, the ratios of the concentrations of diazoxon to diazinon ranged from 0 to 0.5. Diazinon and diazoxon were sometimes detected in the same samples and sometimes did not co-occur in samples.

Figure 3-5. Diazinon concentrations in surface water between 1991 and 2013 based on NAWQA, STORET, and CADPR Data.

The figure is shown with a log scale and without a log scale. In the graph without the log scale, six data points above 10 µg/L are not displayed.

Figure 3-6. Number of surface water samples collected and analyzed for diazinon

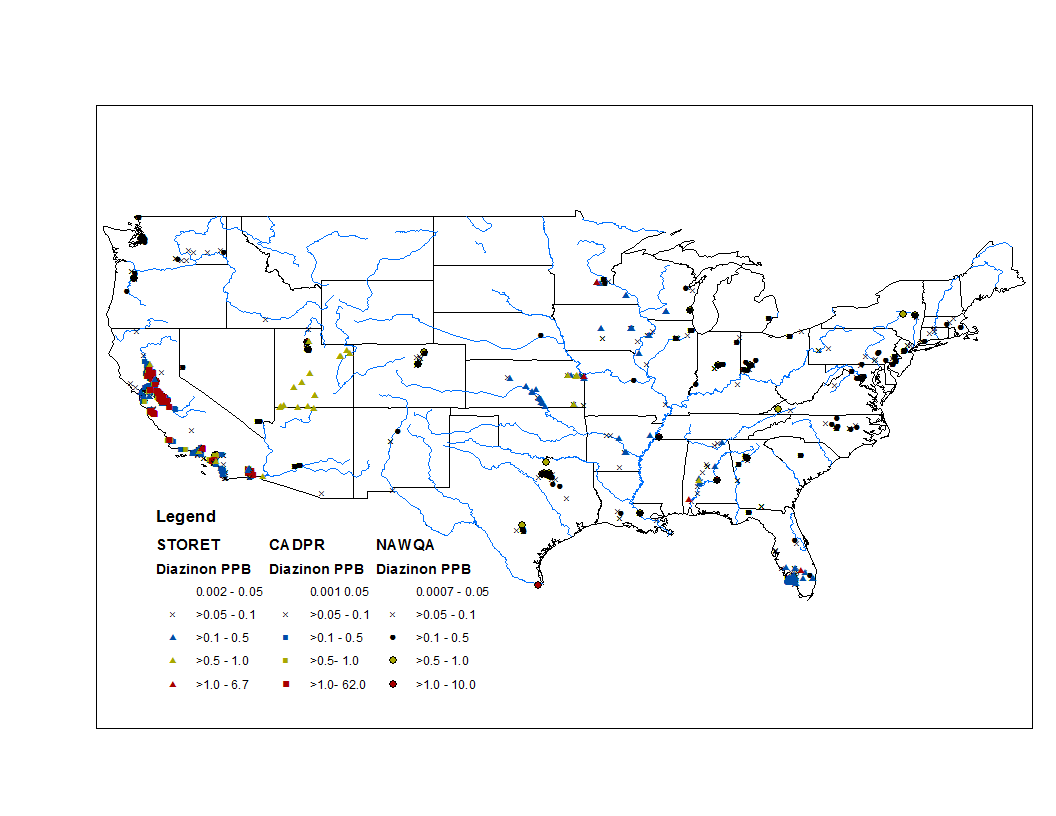


Figure 3-7. Diazinon concentrations in surface water in µg/L (ppb) across the United States based on data obtained from STORET, NAWQA, and CADPR

Table 3-18. Summary of surface water monitoring data

| **Sites (Dataset Source)** | **Year** | **Study Type** | **Sampling Frequency** | **Maximum Diazinon Conc. µg/L** | **Detection frequency**  **(Detects/samples)** | **Source** |
| --- | --- | --- | --- | --- | --- | --- |
| **Diazinon** |  |  |  |  |  |  |
| National (NAWQA) | 1993 - 2014 | General | Irregular | 3.8  (0.359 after 2003) | 27%  (8313/30,297) | NAWQA  (USGS, 2015b) |
| 12 Drinking Water Reservoirs (USGS/USEPA) | 1999-2000 | Collected in areas of high pesticide use | Quarterly and weekly to biweekly during pesticide use season | 0.11 | 35%  (114/323) | USEPA and USGS  (Blomquist *et al.*, 2001) |
| Raw and finished drinking water across the U.S. (USDA) | 2001-2013 | General | Irregular | 0.133 | 0.10%  (6/5,921) | PDP  (USDA, 2013) |
| National (STORET) | 1986-2012 | General | Irregular | 6.7 | 8%  (1784/22,616) | STORET  (USEPA, 2015) |
| South Florida | 1992-2007 | General | Quarterly | 1.9 | 21%  (15/71) | (Pfeuffer, 2011) |
| Washington Cranberry Growing Area | 1994 – 2012 | Collected in cranberry drainage ditch pre and post pesticide application | Every 2 days with a total of 5 samples from 2 ditches | 7.0 | 56 to 100% of samples, depending on the site | (Anderson and Davis, 2000; Baker, 2014) |
| Washington State | 1992 – Present | General | Irregular | 5.7 | 5% (233/4,667) | (Washington State Department of Ecology, 2015) |
| Oregon | 1998 | General | Irregular | -- | 0% (0/190) | (Oregon Department of Environmental Quality, 2015) |
| **California Data Analysis** | | | | | | |
| California (CADPR) | 1990-2012 | General | Irregular | 61.9  (in 2009) | 33%  (4495/13,620) | (CADPR, 2012) |
| California (CEDEN) | 1993-2014 | General | Irregular | 6.7  1.15  (after 2007) | 47%  (1,680/3,563) | (State Water Resources Control Board, 2015) |
| Regions of California with High Detection Frequency (CADPR) | 2005-2010 | General | Irregular | 24 | 10 – 91% | (Zhang *et al.*, 2012) |
| California, irrigation season use (CADPR) | 2003-2008 | Filtered data for areas not influenced by dormant and urban use and analyzed diazinon use data | Irregular | 9% exceeded 0.16 (Max not reported) | 24%  (637/2,635) | USGS and CADPR  (Starner, 2009) |
| San Joaquin River (USGS) and Tributaries | January-February 2000 | Collected during dormant season for orchards in known diazinon use area | Weekly during dry periods and more than weekly during wet periods | 1.06 | 82-100% of samples depending on the site | (Kratzer *et al.*, 2002) |
| January-February 2000 | 0.435 | 95-100% of samples depending on the site | (Zamora *et al.*, 2003) |
| April – August 2001 | 12 sites sampled in areas receiving drainage from orchards and field crops | Weekly between April and August | 0.325 | 10-100% of samples depending on the site | (Domagalski and Munday, 2003) |
| Sacramento River and Tributaries (USGS) | January – February 2000 | Collected during dormant season for orchards | 5 consecutive days after 3 storm events | 2.89 | 77%  (106/138) | (Dileanis *et al.*, 2002) |
| January – February 2001 | 5 consecutive days after two storm events | 1.38 | 0-100%, depending on site | (Dileanis *et al.*, 2003) |
| Santa Clara River and Callequas Creek Watersheds | Wet and dry season 2009 | Collected during wet season after rain events and during dry season | One sample collected at 14 sites after two rain events and 2 samples collected during dry season | 0.172 | 82% during wet season 4% during dry season | (Delgado-Moreno *et al.*, 2011) |
| Salinas River, where agricultural drains enter river | 2000-2001 | General | 4x in 2 years | 3.340 | 44%  (17/39) | (Anderson *et al.*, 2003) |
| Central Coast California Monitoring Data | 2006 to 2013 | General | Irregular | 24.46 | 37%  (80/216) | Central Valley Water Quality Control Board (email dated 2/26/2015) |
| Central Coast California Monitoring Data | 2000-2011 | General | Irregular | 40.8 | 34%  (3024/8963) | Central Valley Water Quality Control Board (email dated 1/29/2015) |
| Central Valley of California TMDL (UCDavis) | Winter 2006 | Counties with known diazinon use | Daily for 2 to 8 days following storm event | 0.778 | 50-100% | (Regional Water Quality Control Board, 2006) |
| **Diazoxon** |  |  |  |  |  |  |
| NAWQA National | 2002 – 2014 | General | Irregular | 0.06 | 2%  (30/1499) | NAWQA  (USGS, 2015b) |
| National (STORET) | 2009, 2012-2013 | General | Irregular | Below LOQ (0.075 to 0.15 µg/L) | 8%  (10/2900) | STORET  (USEPA, 2015) |
| California (CADPR) | 1991-1995 | General | Irregular | 0.43 | 0.6%  (5/773) | CADPR Surface Water Protection Program Database (CADPR, 2012) |

LOQ=Limit of Quantitation

a Field study refers to sampling occurring after a known pesticide application at a known location, with a well-described relationship to the sampling event. General refers to studies in which, when samples were collected, no consideration was given to pesticide use patterns.

##### Pesticide Concentrations in Drinking Water USGS and USEPA in (1999-2000)

In 1999 and 2000, the United States Geological Survey (USGS) and USEPA collaborated in examining concentrations of pesticides in twelve small drinking-water supply reservoirs in areas of high pesticide use. The reservoirs range in size from 120 to 92,600 acre-feet with contributing watersheds ranging in size from 3.3 to 784 square miles (Blomquist *et al.*, 2001). Water samples were collected from raw-water intakes, finished drinking water, and some reservoir outflows. Samples were collected quarterly throughout the year and at weekly or biweekly intervals following the primary pesticide application periods. Diazinon was detected in 35% (114 of 323) of raw water samples and was one of the most frequently detected insecticides, with a maximum concentration of 0.11 µg/L detected in Lake Arcadia, Oklahoma. This was the reservoir with the smallest capacity among those sampled (120 acre-feet) and a high sampling frequency. Its watershed includes both urban and agricultural land uses. Diazinon was not detected in any of the finished water samples. Diazoxon was not included as an analyte in this study. Other studies have shown that organophosphorus insecticides are readily oxidized in the presence of chlorine, suggesting that diazoxon could form (see Drinking Water Treatment Section). Although diazinon was not observed in finished water samples, it is possible that diazoxon was present.

##### Pesticide Data Program (PDP) Surface Water (2001-2013)

The Pesticide Data Program (PDP) is a national pesticide residue database program that examines pesticide residues in agricultural commodities and drinking water in the United States’ food supply, to support pesticide dietary exposure assessments (USDA, 2013). Finished drinking water monitoring in California and New York began in 2001. In 2002, the program was expanded to Colorado, Kansas, and Texas. In 2004, the program began examining paired raw and finished drinking water samples sourced from surface water. The survey ended in 2013. The limit of detection ranged from 3.3 to 30 ng/L.

Diazinon was detected in 0.10% of surface water source water samples (six of 5,921 samples) at a maximum concentration of 0.133 µg/L (**Table 3-19**). Detections occurred in 2001, 2002, 2003, 2007, and 2010. Most detections were in raw water; however, there were some detections in finished water.

Table 3-19. Summary of surface water sourced drinking water monitoring data from the PDP

| **Year** | **Detects** | **Number of Samples** | **Frequency of Detects** | **Diazinon Max Concentration (µg/L)** | **Detect(s) in** |
| --- | --- | --- | --- | --- | --- |
| 2001 | 1 | 283 | 0.35% | 0.010 | Finished Water |
| 2002 | 1 | 657 | 0.15% | 0.010 | Finished Water |
| 2003 | 1 | 794 | 0.13% | 0.133 | Finished Water |
| 2004 | 0 | 239 | 0.00% | NA | Paired raw and finished water |
| 2005 | 0 | 232 | 0.00% | NA | Paired raw and finished water |
| 2006 | 0 | 368 | 0.00% | NA | Paired raw and finished water |
| 2007 | 1 | 733 | 0.14% | 0.0164 | Paired raw and finished water |
| 2008 | 1 | 619 | 0.16% | 0.1 | Paired raw and finished water |
| 2009 | 0 | 612 | 0.00% | NA | Paired raw and finished water |
| 2010 | 1 | 559 | 0.18% | 0.059 | Paired raw and finished water |
| 2011 | 0 | 240 | 0.00% | NA | Paired raw and finished water |
| 2012 | 0 | 485 | 0.00% | NA | Paired raw and finished water |
| 2013 | 0 | 100 | 0.00% | NA | Paired raw and finished water |
| Total | 6 | 5921 | 0.10% | 0.133 | Raw and finished water |

NA=not applicable

#### Groundwater

Diazinon has also been detected in groundwater, though at a lower frequency (0 to 3%) than in surface water, and typically at lower concentrations (**Table 3-.20**). Although the maximum groundwater detection was 19 µg/L, the majority of detections were at lower concentration. Detections occurred in Colorado, Idaho, Iowa, California, Connecticut, Florida, Illinois, Indiana, Louisiana, Maryland, Massachusetts, Michigan, Minnesota, Nevada, New Hampshire, New Mexico, New York, North Carolina, Pennsylvania, South Carolina, and Virginia.

Table 3-20. Summary of groundwater monitoring data

| **Sites (Dataset Source)** | **Year** | **Study Type** | **Sampling Frequency** | **Maximum Diazinon Conc. µg/L** | **Detection frequency**  **(Detects/samples)** | **Source** |
| --- | --- | --- | --- | --- | --- | --- |
| **Diazinon** |  |  |  |  |  |  |
| National (NAWQA) | 1992-2014 | General | Varies | 19  (0.098 after 2002) | 0.8%  (105/12,640) | (USGS, 2015a) |
| Private Drinking Water Wells on farms, schools, daycares across the nation and municipal drinking water (USDA) | 2007-2013 | General | Varies | 0.081  (in 2013) | 0.16%  (3/1,915) | PDP  (USDA, 2013) |
| Private wells in vulnerable counties in New York | 2007-2009 | Vulnerable private wells in rural areas | Single Sample collected | 0.1 | 3%  (2/80) | (Richards *et al.*, 2012) |
| Oregon | 1998 | General | Varies | -- | 0% (0/71) | (Oregon Department of Environmental Quality, 2015) |

#### Sediment and Tissue

Diazinon has also been detected in sediment, at a maximum concentration of 0.46 µg/L in pore water and 4.72 µg/kg-dry weight sediment (**Table 3-21**). The frequency of detection in sediment is much lower than that in water, ranging from 1 to 60 percent of samples among data sources.

Table 3-21. Summary of sediment and tissue monitoring data

| **Sites (Dataset Source)** | **Year** | **Study Type** | **Sampling Frequency** | **Maximum Diazinon Concentration** | | | **Source** |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Pore water**  **µg/L** | **Sediment**  **µg/kg-dw** | **Detection frequency**  **(Detects/samples)** |
| National (NAWQA) | 1992-2007 | General | Varies | -- | 3.5 | 1%  (3/242) | (USGS, 2015a) |
| Oregon | 1998 | General | Varies | -- | 8 | 20% (1/5) | (Oregon Department of Environmental Quality, 2015) |
| Salinas River, where agricultural drains enter river | 2000-2001 | General | 4x in 2 years | 0.46 | -- | 44% (3/9) | (Anderson *et al.*, 2003) |
| Central Coast California Monitoring Data | 2006 to 2009 | General | Varies | 0.03 | 4.72 | 9% (11 of 122) | Central Valley Water Quality Control Board (email dated |
| Santa Clara River and Callequas Creek Watersheds | Wet and dry season 2009 | Collected during wet season after rain events and during dry season | One sample collected at 14 sites after two rain events and 2 samples collected during dry season |  | Median=1 ng/g | 60% | (Delgado-Moreno *et al.*, 2011) |

Tissue data were obtained from the California Environmental Data Exchange Network (CEDEN) on January 10, 2015. Data on tissue containing residues of diazinon were reported by the Surface Water Ambient Monitoring Program, the Regional Monitoring Program for Water Quality, and the Newport Bay Watershed Biotrend Monitoring Program.

Twenty detections were reported between 1984 and 1989 on residues in freshwater clams (*Corbicula fluminea*) and in California Mussels (*Mytilus californianus*). Diazinon was present at concentrations ranging from 1,060 ng/g-lipid to 13,853.4 ng/g-lipid. Samples were collected from rivers, creeks, harbors, canals, and sloughs.

There were 166 detections in freshwater clam, California Mussel, Sailfin Molly (*Poecilia latipinna*), Asiatic clam (*Corbicula manilensis*), Channel catfish *(Ictalurus punctatus*), Common carp (*Cyprinus carpio*), Fathead minnow (*Pimephales promelas*), goldfish (*Carassius auratus*), red shiner (*Cyprinella lutrensis*), Treespine stickleback (*Gasterosteus aculeatus*), longjaw mudsucker (*Gillichthys mirabilis*), Tilapia spp., mosquitofish (*Gambusia affinis*), white croaker (*Genyonemus lineatus*), red rock crab (*Cancer productus*), and Jacksmelt (*Atherinopsis californiensis*). Detected concentrations were a maximum of 1100 ng/g dry-weight (usually whole organisms without gut but some soft tissue) and 1050 ng/g wet-weight whole organism. The highest concentration reported in fillet was 140 ng/g wet-weight.

#### Atmospheric Monitoring

Diazinon is one of the most frequently detected pesticides in air and in precipitation. The majority of monitoring studies involving diazinon have been conducted in California; however, diazinon has been detected throughout the United States in air and precipitation. Available air and precipitation monitoring data for diazinon in California are reported in **Table 3-22**.

The magnitude of detected concentrations of diazinon in air and in precipitation could vary based on several factors, including proximity to use areas and timing of applications. In air, diazinon has been detected at concentrations up to 0.306 µg/m3. Measured concentrations of diazinon in rain in California have been detected at concentrations up to 2.22 µg/L. In fog, diazinon has been detected at up to 76.3 µg/L (Majewski and Capel, 1995). Deposition studies in California in diazinon use areas show that diazinon was detected in rain after applications to orchards. Wet deposition generally had higher concentrations of diazinon than dry deposition. Diazinon was detected in 93% of rain samples (n=137), with mean and maximum concentrations of 0.149 and 2.220 µg/L, respectively (Majewski *et al.*, 2006). Diazinon has been detected in California lakes (maximum concentration of 0.0741 µg/L) that do not receive runoff or spray drift from agricultural areas and are presumed to receive inputs of diazinon from atmospheric deposition only (Fellers *et al.*, 2004; LeNoir *et al.*, 1999).

Zabik and Seiber (1993) collected air samples in 1990 and 1991 from a national park in the Sierra Nevada Mountains and analyzed for both diazinon and diazoxon. The authors reported that 26% of the parent was converted to the oxon in air. In paired air samples, the ratio of the oxon to the parent ranged 0.068-3.9 (N = 34). Zabik and Seiber (1993) also collected rain samples deposited in the Sierra Nevada Mountains. In their limited samples, they detected diazoxon at lower levels compared to the parent.

Diazinon and diazoxon have been quantified in fog in two different studies conducted in California. In 1986, diazinon concentrations ranged 0.31-18 µg/L and diazoxon concentrations ranged 0.42-28 µg/L (n = 6). The ratios of the oxon to the parent ranged 0.056-7.1, where the majority of the samples had concentrations of the two that were on the same order of magnitude (Glotfelty *et al.* 1990). In 1987, diazinon concentrations ranged 0.15-4.8 µg/L and diazoxon concentrations ranged 1.9-11 µg/L (n = 5). The ratios of the oxon to the parent ranged 0.067-13, where the majority of the samples had concentrations of the two that were on the same order of magnitude (Schomburg *et al.* 1991).

Table 3-22. Diazinon detections in air and precipitation samples taken in California

| **Location** | **Year** | **Sample type** | **Maximum Conc.\*** | **Detection frequency** | **Source** |
| --- | --- | --- | --- | --- | --- |
| CA, MD | 1970s-1990s | Air | 0.306 | NA | (Majewski and Capel, 1995) |
| Sequoia National Park, CA | 1996 | Air | 0.00024 | 41.7% | (LeNoir *et al.*, 1999) |
| Sacramento, CA  (Franklin Field Airport) | 1996-1997 | Air | 0.0191 | 37.1 % | (Majewski and Baston, 2002) |
| Sacramento, CA (Sacramento Metropolitan Area) | 1996-1997 | Air | 0.0122 | 46.5 % | (Majewski and Baston, 2002) |
| Sacramento, CA (Sacramento International Airport) | 1996-1997 | Air | 0.112 | 38.5 % | (Majewski and Baston, 2002) |
| Fresno County, CA | 1997 | Air | 0.290 | NA | (State of California, 1998a) |
| Fresno County, CA | 1998 | Air | 0.160 | NA | (State of California, 1998b) |
| Mississippi River from New Orleans, LA to St. Paul MN | 1994 | Air | 0.00036 | 100% | (Majewski *et al.*, 1998) |
| Central Valley, CA | 1990-1991 | Air | 0.01 (parent)  0.003 (diazoxon) | 100% | (Zabik and Seiber, 1993) |
| Orchard Application in Glenn county California | 01/2010 | Air | 4.261 (parent)  0.124 (diazoxon) | 85% | (Rider, 2010) |
| 3 California Agricultural Communities (Salinas, Shafter, Ripon) | 2014 | Air | 0.0057 (diazoxon) | 0%(parent)  1%(diazoxon) | (Tuli *et al.*, 2015) |
| 3 California Agricultural Communities (Salinas, Shafter, Ripon) | 2013 | Air | 0.0487 (parent)  0.0258 (diazoxon) | 4% (parent)  1% (diazoxon) | (Vidrio *et al.*, 2014) |
| 3 California Agricultural Communities (Salinas, Shafter, Ripon) | 2012 | Air | 0.0052 (parent)  0.0101 (diazoxon) | 3% (parent)  3% (diazoxon) | (Vidrio *et al.*, 2013b) |
| 3 California Agricultural Communities (Salinas, Shafter, Ripon) | 2011 | Air | 0.0596 (parent)  0.036 (diazoxon) | 13% (parent)  8% (diazoxon) | (Vidrio *et al.*, 2013a) |
| Sequoia national Park, CA | 1995-1996 | Rain | 0.019 | 57 % | (McConnell *et al.*, 1998b) |
| San Joaquin River Basin, CA | 2001 | Rain | 0.908 | 100% | (Zamora *et al.*, 2003) |
| San Joaquin Valley, CA | 2002-2004 | Rain | 2.22 | 93% | (Majewski *et al.*, 2006) |
| Central Valley, CA | 1990-1991 | Rain | 6.1 (parent)  2.3 (diazoxon) | 100% | (Zabik and Seiber, 1993) |
| CA, MD | 1970s-1990s | Fog | 76.3 | NA | (Zhang *et al.*, 2012) |
| Parlier, CA | 1986 | Fog | 18.0 | NA | (Glotfelty *et al.*, 1990) |
| Monterey, CA | 1987 | Fog | 4.80 | NA | (Schomburg *et al.*, 1991) |
| Sequoia national Park, CA | 1995-1996 | Snow | 0.014 | 62.5 % | (McConnell *et al.*, 1998a) |

\*For Air, µg/m3; for rain, snow and fog, µg/L

## Monitoring Results

### WARP Model and Extrapolation of Monitoring Results

The Watershed Regression for Pesticides for multiple pesticides (WARP-MP) Map Application recently became available on the US Geologic Services website (<http://cida.usgs.gov/warp/home/>). The WARP models for pesticides are developed using linear regression methods to establish quantitative linkages between pesticide concentrations measured at NAWQA and National Stream Quality Accounting Network (NASQAN) sampling sites and a variety of human-related and natural factors that affect pesticides in streams. Such factors include pesticide use, soil characteristics, hydrology, and climate - collectively referred to as explanatory variables. Measured pesticide concentrations, together with the associated values of the explanatory variables for the sampling sites, comprise the model-development data.

The WARP-MP Map Application is built upon the atrazine WARP models, in conjunction with an adjustment factor for each pesticide. The WARP model for estimating atrazine in streams is based on concentrations measured by NAWQA and NASQAN from 1992 to 2007 at 114 stream sites. The atrazine model actually consists of a series of models, each developed for a specific concentration statistic (annual mean and 4-, 21-, 30-, 60-, and 90-day annual maximum moving average). The models are built using the explanatory variables that best correlate with, or explain, the concentration statistics computed from concentrations observed in streams. Although explanatory variables included in the models are significantly correlated with pesticide concentrations, the specific cause-and-effect relations responsible for the observed correlations are not always clear, and inferences regarding causes should be considered as hypotheses.

The WARP models used on the Map Application web site to create maps and graphs are the models for the annual mean and annual maximum moving averages (4-, 21-, 30-, 60-, and 90-day durations). For each of these annual concentration statistics, the models can be used to estimate the value for a particular stream, including confidence bounds on the estimate, or the probability that a particular value will be exceeded, such as a water-quality benchmark. Each of these options for applying the model has advantages for specific purposes. It’s also important to recognize that averaging concentrations over several days or weeks can mask peak concentrations of shorter durations that are toxicologically relevant.

Model estimates are made for most stream reaches included in the U.S. Environmental Protection Agency (EPA) River Reach file (USGS, <http://water.usgs.gov/lookup/getspatial?erf1_2>), which include more than 600,000 miles of streams and more than 60,000 individual stream reaches with watersheds. The flowrates for the streams found in the River Reach file are comparable to those for the flowing aquatic bins (median flowrate of 1.81 m3/s, range 2x10-5 – 18,000 m3/s, n=59,941). The term "stream" refers to all River Reach file segments, regardless of drainage basin area. Model development stations spanned over 5 orders of magnitude in terms of watershed area and model predictions are not biased with respect to watershed area. However, model estimates are not made available for streams with watersheds smaller than 75 square kilometers because of the high potential for uncertainty in explanatory variables, such as pesticide use, for small watersheds. In addition, model estimates are not made available for streams with watershed characteristics, including pesticide-use intensity, outside the range of those used to develop the WARP models.

Annual agricultural pesticide use intensity was estimated from USGS county-level pesticide use estimates (<http://water.usgs.gov/nawqa/pnsp/usage/maps/>). The county-level pesticide use estimates for the U.S. were then combined with national land cover data to generate a raster dataset of pesticide use intensity. When used to estimate the value of a concentration statistic for a stream, such as the annual mean, the model computes the median estimate of the statistic for all streams with watershed characteristics that are similar to the stream in question. Thus, the computed estimate for a particular stream has an equal chance of being above or below the actual value of the statistic. The confidence that the estimated value is within a certain magnitude of the actual value is indicated by the 95-percent confidence limits, which encompass 95 percent of the actual values associated with the predicted value.

For 2012, **Table 3-16** provides the range of the estimated 4-day moving average concentrations and the upper bound 4-day moving average concentrations for diazinon by HUC2. The 4-day averages are reported, as peak concentrations are not provided by the Map Application. These values are on the low end and below the range of PRZM5/VVWM maximum 1-in-15 year 4-day average concentrations modeled in bin 7 and an order of magnitude or more lower than the 1-in-15 year 4-day average EECs modeled in the other aquatic bins.

Table 3-16. WARP Map Application estimated 4-day moving average concentrations for diazinon

| **HUC 2** | **Count of Detects (Total Count)** | **Range of Estimated 4-day Moving Average Concentrations (µg/L)** | **Range of Upper Bound 4-day Moving Average Concentrations (µg/L)** |
| --- | --- | --- | --- |
| 1 | 303 (305) | <0.001 - 0.005 | <0.001 - 0.169 |
| 2 | 759 (1022) | <0.001 - 0.106 | <0.001 - 3.905 |
| 3 | 454 (767) | <0.001 - 0.071 | <0.001 - 2.824 |
| 4 | 143 (215) | <0.001 - 0.020 | <0.001 - 0.785 |
| 5 | 355 (439) | <0.001 - 0.014 | <0.001 - 0.540 |
| 7 | 21 (21) | <0.001 | <0.001 |
| 8 | 114 (114) | <0.001 | <0.001 |
| 10 | 434 (434) | <0.001 | <0.001 |
| 11 | 553 (522) | <0.001 - 0.031 | <0.001 - 1.236 |
| 12 | 1000 (1336) | <0.001 - 0.122 | <0.001 - 4.893 |
| 13 | 241 (316) | <0.001 - 0.060 | <0.001 - 2.393 |
| 14 | 445 (479) | <0.001 - 0.006 | <0.001 - 0.220 |
| 15 | 426 (440) | <0.001 - 0.016 | <0.001 - 0.644 |
| 16 | 282 (301) | <0.001 - 0.010 | <0.001 - 0.363 |
| 17 | 2162 (2624) | <0.001 - 0.070 | <0.001 - 2.551 |
| 18 | 284 (642) | <0.001 - 0.100 | <0.001 - 3.837 |

There are a number of uncertainties and limitations that accompany the use of the WARP-MP model. Application of the regression models to streams with watershed parameters outside the range of the sites used to develop the models will result in increased uncertainty. In particular, application of the models to lakes or reservoirs would likely result in variably biased predictions because the models were developed with data from flowing water systems. Furthermore, the sampling frequencies of the model-development sites may not be sufficient to reliably characterize the highest moving-average concentrations during a year, particularly for the shorter-duration averages. For some streams, the short-duration moving-average concentrations used in model development may be underestimated by as much as 84%, as estimated for atrazine. Thus, application of the models to predict the maximum moving-average concentrations for short durations, such as the 4-d moving-average, is expected to generally underpredict the actual concentrations for pesticides that performed well with the model development and evaluation sites. Application of these models to pesticides that are used more heavily during months that differ from the high pesticide runoff period months used in model development may result in increased uncertainty and potentially biased results. Also, the pesticide-use intensity variable was limited to agricultural-use intensity because use data for other purposes were not available. Application of these models to pesticides with significant nonagricultural use may result in increased uncertainty and potentially biased results. Applications of the WARP-MP model to pesticides with properties outside the range of those used in development of the adjustment factor have greater uncertainty.

## Aquatic Exposure Summary

Modeled EECs represent an upper bound on potential exposure as a result of the use of diazinon. Modeled 1-in-15-year peak EECs for bins 2, 5, 6, and 7 ranged from 0.75 to 1615x ambient monitored concentrations. As recommended by the NRC in the 2013 NAS report, ambient 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 present a lower bound on known exposure.

## Uncertainties in Aquatic Modeling and Monitoring Estimates

### Surface Water Aquatic Modeling

Exposure to aquatic organisms from pesticide applications is estimated using PRZM/VVWM EECs. Regional differences in exposure are assessed using regionally-specific PRZM scenarios (*e.g.*, information on crop growth and soil conditions) and meteorological conditions at the HUC 2 level (**Section 2.3 Scenario Selection**). The information used in these scenarios is designed to reflect conditions conducive to runoff. In instances where PRZM 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 VVWM 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 VVWM 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. Watershed areas are developed using NHDPlus data for each HUC 2 region and a log-log regression of drainage area to flowrate. Where contributing watershed areas are smaller than those predicted, this would result in less mass loading and runoff contributions to the waterbody and lower concentrations

The assessment relies on maximum use patterns (**Section 2 Measures of Aquatic Exposure**). In situations where use patterns are less than the labeled maximums, environmental exposures will be lower.

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

The aquatic modeling assumes a constant wind of 10 mph blowing directly toward the waterbody (**Section 2.4.2 Spray Drift**). These assumptions are conducive to drift transport and result in maximum potential loading to the waterbody. However, in many situations the wind will not be blowing constantly and directly toward the water body at this speed; therefore, aquatic deposition will likely be less than predicted. Additionally, many labels and applicator best management practices encourage not applying pesticides when the wind is blowing in the direction of sensitive areas (*i.e.*, listed species habitat). Lastly, reductions in spray drift deposition due to air turbulence, interception of spray drift on nearby plant canopy, and applications during low wind speeds are not taken into account in the spray drift estimates; therefore, loading due to spray drift may be over-estimated.

There is uncertainty associated with the selection of PRZM/VVWM input parameters. In this regard, one of the important parameters that can impact concentration estimates is the selection of application dates (**Section 2.4.3 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 PRZM/VVWM models use 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.

In the case of applications to rice paddies or cranberry bogs, the PFAM model is used to estimate concentrations in the flooded field (**Section 2.5.2 Cranberry Modeling**). For listed species that may visit a paddy or bog, the water column and sediment estimates are intended to be protective of exposures they may encounter. However, for listed species whose habitat is outside the flooded area, the use of water column and sediment concentrations from the paddy or bog is likely to overestimate exposure due to dilution and dispersion of the pesticide when discharged to a flowing waterbody. Additionally, as these flooded field systems tend to have water management controls which regulate the maximum release of paddy or bog water, the pesticide concentration in the water, and allow for manual releases in the fall during harvest, exposure could be limited to a specific time of year and not year-round.

#### Aquatic Bins 3 and 4

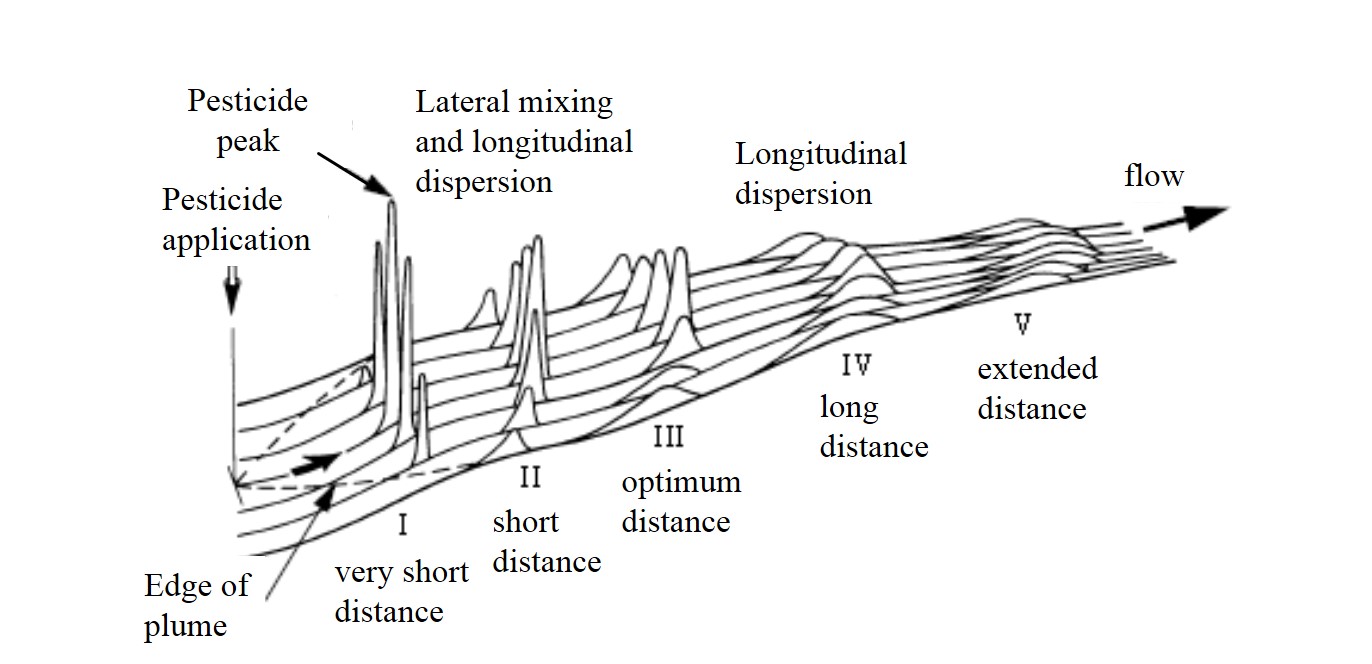
PRZM/VVWM are a field-scale models. Flowing water bodies such as streams and rivers with physical parameters consistent with aquatic Bins 3 and 4 have watershed areas well beyond those of typical agricultural fields. Watershed sizes assumed for Bin 3 habitats exceeded 10,000 acres and Bin 4 watersheds were assumed to be greater than 4 million acres. Initial modeling efforts, applying the field-scale model to these large watersheds and using the same scenario parameters as those used for the other bins, results 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 3-8**), 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). Several adjustments, discussed in **ATTACHMENT 3-1 Background Document: Aquatic Exposure Estimation for Endangered Species**, have been made to the inputs and outputs to reflect changes that would be anticipated in modeling such scenarios. 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 3-8. Effect of Pesticide Concentration via Advective Dispersion

#### Monitoring Data

While general monitoring data may indicate lower levels of the active ingredient in surface water, general monitoring is typically not focused in use areas or conducted during times of known use. As the delineated by NAS (2013), general monitoring studies “are not associated with specific applications of pesticides under well-described conditions” and “cannot be used to estimate pesticide concentrations after a pesticide application or to evaluate the performance of fate and transport models.” Additionally, sampling intervals for general monitoring datasets are sporadic and as such may not reflect potential peak concentrations that may occur in surface waters when runoff events occur shortly after application. Adding to this uncertainty, reporting limits for some of the datasets have varied over the years. Although monitoring datasets may report trace levels (*e.g.*, levels above the method detection limit but below the reporting limit), consistent detection limits would facilitate interpretation of the monitoring results from year to year and across datasets. As a result, general monitoring data are used in this assessment as part of the weight of evidence analysis to present a lower bound on known exposure.

The preceding discussion outlined the uncertainties associated with the modeling techniques used to derive EECs, particularly the methods used to derive EECs for the moderate and high flowing aquatic bins, bins 3 and 4. Alternative recommendations from stakeholders, the scientific community, and the public at large on how to estimate pesticide exposure in these waterbodies on a watershed scale or to improve the proposed modeling methodology are encouraged.

# Measures of terrestrial exposure

## Introduction

Terrestrial animals may be exposed to diazinon through multiple routes of exposure, including diet, drinking water, dermal and inhalation. If the species consumes plants, invertebrates or vertebrates (amphibians, reptiles, birds or mammals) that inhabit terrestrial areas, T-REX is used by EFED. If the species consumes aquatic organisms, then KABAM is used. As noted in the Problem Formulation, to improve efficiency and expand EFED’s modeling capabilities to other, non-dietary routes of exposure for terrestrial organisms, the Terrestrial Effects Determination (TED) tool was developed. This tool integrates T-REX, T-HERPS and the earthworm fugacity model, along with several other models used by EFED. When this document indicates that T-REX or the earthworm fugacity models should be run for a species, the TED tool will be run. Assessors could also run the current version of T-REX. As discussed in the terrestrial exposure appendix, KABAM will not be run for chlorpyrifos, diazinon or malathion. In its place, BCF values will used to estimate exposure through consumption of aquatic food items. The spray drift model, AgDRIFT, will be used in the effects determinations to characterize the distance from the edge of the field to which exposure is at levels of concern for a species.

Two major parameters are used in Tier I modeling to represent 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 in **ATTACHMENTS 1-16 THRU 1-19**.

This section characterizes the estimated exposures of diazinon on different food items in the terrestrial environment and in fish (which may be consumed by piscivorous mammals and birds). These values are used to generate dose-based dietary exposure estimates. Species specific dose-based exposures through diet, drinking water, dermal and inhalation routes will be provided in the TED tool outputs. **ATTACHMENT 1-7** discusses the methods for estimating dose-based exposures. Upper bound exposure estimates are used in Step 1 of the ESA process, with upper bound and mean residues over time being used in Step 2.

Four different diazinon application scenarios were used to estimate terrestrial exposure: 1) a minimum single application rate (0.5 lb a.i./A); 2) an upper-bound single application rate (3.0 lb a.i./A); 3) a maximum single application rate (4.0 lb a.i./A); and 4) a multiple application scenario (3 applications at 3 lb a.i./A with 14 d intervals). These application scenarios are meant to be representative of the range of application rates and uses for diazinon. The first two scenario are based on the range of single application rates allowed for foliar applications to orchards, ground fruit and vegetables and nurseries. The third rate is based on the maximum application rate allowed for diazinon, which is for spray applications to soil. The fourth application scenario represents the maximum application scenario allowed (maximum number of applications at the highest rate), which applies to cranberries.

## Estimated concentrations in terrestrial food items (mg a.i./kg-food)

The TED tool generates estimates of pesticide concentrations in above-ground terrestrial invertebrates, grass (tall and short), broadleaves, fruit and seeds. Recent additions to the model have allowed for calculation of pesticide concentrations in soil-dwelling invertebrates (using earthworm partitioning model) and in terrestrial vertebrates (*i.e.*, birds and mammals; using the T-HERPS model).

The T-REX model is intended to simulate foliar spray applications of pesticides. **Table 3-23** summarizes the maximum foliar spray application rates registered for diazinon. Single maximum foliar application rates range 0.5-3.0 lb a.i./A. In order to bound exposure estimates, minimum and maximum application scenarios are used to run T-REX. The minimum scenario is one application of 0.5 lb a.i./A, which is represented by applications to figs, filbert (hazelnut), ginseng and lettuce. The maximum application scenario is for cranberries, where a maximum rate of 3 lb a.i./A may be applied 3 times per season (with a minimum retreatment interval of 14 day). T-REX accounts for dissipation of pesticide residues on food items. A foliar dissipation half-life of 5.3 days is used for diazinon, based on data reported by Willis and McDowell (1987).

Table 3-23. Diazinon maximum single foliar application rates, number of applications per crop (season) and interval between applications.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Crop** | **Application method** | **Single application rate (lb a.i./A)** | **Number of applications** | **Interval between applications (d)** |
| Figs, filbert | Ground, airblast | 0.5 | 1 | none |
| Ginseng | Ground | 0.5 | 1 | none |
| Lettuce | Ground, aerial | 0.5 | 1 | none |
| Peas | Ground | 0.5 | 3 | Not available |
| Peppers, swiss chard, cucumbers | Ground | 0.5 | 5 | 7 |
| Turnips | Ground | 0.5 | 5 | 3 |
| Squash | Ground | 0.75 | 5 | 7 |
| Melons | Ground | 0.8 | 1 | none |
| Tomatoes | Ground | 0.8 | 5 | 7 |
| Outdoor ornamentals (nurseries) | Ground, airblast | 1 | 1\* | none |
| Strawberries | Ground | 1 | 1 | none |
| Blueberries | Ground, airblast | 1 | 2 | 30 |
| Pineapples | Ground, airblast | 1 | 2 | 28 |
| Parsnips | Ground | 1 | 5 | 7 |
| Caneberries | Ground, airblast | 2 | 1 | none |
| Apples | Ground, airblast | 2 | 2 | 14 |
| Cherries | Ground, airblast | 2 | 2 | 30 |
| Stone fruit (apricots, cherries, peaches, nectarines, plums, prunes) | Ground, airblast | 2 | 2 | 60 |
| Pears | Ground, airblast | 2 | 2 | 70 |
| Almonds | Ground, airblast | 3 | 1 | none |
| Cranberries | Ground, airblast | 3 | 3 | 14 |

\*per season. Multiple applications are allowed.

Diazinon is also registered for ground spray applications to soil with incorporation. A single maximum application of 4 lb a.i./A may be made to vegetable and fruit field crops (*e.g*., melons, peppers, potatoes). EECs are generated using T-REX for terrestrial invertebrates (above ground) and soil-dwelling invertebrates in order to estimate diazinon concentrations in food items located on the field or directly adjacent. Although substantial vegetation is not expected to be present on the treated field for the soil application, T-REX EECs were also generated for leaves, fruit and seeds in order to represent pesticide concentrations directly adjacent to the field. **Table 3-24** summarizes the mean and upper bound dietary-based EECs. Additional description of the estimated diazinon concentrations on food items is provided in the following sections.

Table 3-24. Mean and upper bound dietary based EECs calculated for food items consumed by listed birds, terrestrial-phase amphibians or reptiles. Values represent potential exposures for animals feeding on the treated field or in adjacent habitat directly adjacent to the field.1

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Food Item** | **Model** | **Lowest application rate allowed**  **(0.5 lb a.i./A)** | | **Highest foliar application rate allowed**  **(3 applications of 3.0 lb a.i./A)** | | **Highest soil application rate allowed**  **(1 application of 4.0 lb a.i./A)** | |
| **Mean** | **Upper bound** | **Mean** | **Upper bound** | **Mean** | **Upper bound** |
| Terrestrial invertebrates (above ground) | T-REX | 33 | 47 | 195 | 282 | 260 | 376 |
| Terrestrial invertebrates (soil dwelling) | Earthworm fugacity | NA | 9.5 | NA | 57 | NA | 76 |
| Short grass | T-REX | 42.5 | 120 | 255 | 720 | 340 | 960 |
| Tall grass (surrogate for nectar and flowers) | T-REX | 18 | 55 | 108 | 330 | 144 | 440 |
| Broadleaves | T-REX | 23 | 68 | 135 | 405 | 180 | 540 |
| Seeds and fruit | T-REX | 3.5 | 7.5 | 21 | 45 | 28 | 60 |
| Birds (small, insectivore)\*\*\* | T-HERPS | 37 | 54 | 222 | 321 | 296 | 428 |
| Mammals (small, herbivore)\*\*\* | T-HERPS | 40.5 | 114 | 243 | 686 | 324 | 915 |
| Amphibians/reptiles (small, insectivore) | T-HERPS | 1.8 | 2.6 | 11 | 16 | 14 | 21 |
| Aquatic plants | KABAM | 0.28-280\*\* | | | | | |
| Aquatic invertebrates | BCF\* | 0.03-0.82\*\* | | | | | |
| Fish | BCF\* | -720\*\* | | | | | |

1 Additional EECs are available in the TED tool.

\*Based on range of empirical BCFs.

\*\*Varies based on EECs in water.

\*\*\*Also represent residues in carrion.

NA = not applicable

### Terrestrial invertebrates

For terrestrial invertebrates inhabiting the treated field (above ground), upper bound peak EECs range 47-376 mg a.i./kg-food. Mean values range 33-260 mg a.i./kg-food. **Figure 3-9** depicts the estimated concentrations of diazinon on above ground terrestrial invertebrates over time (to aid in reading the figures, the exposure concentrations for the single application of 3.0 lb a.i./A were not shown). When diazinon is applied at a single application of 0.5 lb a.i./A, diazinon upper bound residues are <0.01 mg a.i./kg-food 62 days after the application. For the maximum use scenario (cranberries), upper bound residues of diazinon reach <0.01 mg a.i./kg-food at 108 days after the first application.

Figure 3-9. Mean and upper bound estimated concentrations of diazinon on above ground terrestrial invertebrates.

The earthworm fugacity model was used to estimate pesticide concentrations in soil-dwelling invertebrates located on treated fields. For foliar applications, the steady state concentration of diazinon in soil-dwelling invertebrates is estimated at 14 mg a.i./kg-food for the lower bound application scenario and 84 mg a.i./kg-food for the maximum application scenario. For a single soil application of 4 lb a.i./A, the steady state concentration is 112 mg a.i./kg-food. These values were estimated using a Kd of 6.2 L/kg-soil, which is calculated by multiplying the Koc of 618 L/kg-oc (MRID 49091901) by a fraction of organic carbon (foc) of 0.01. A Log Kow of 3.77 was used. This value is a mean of the 3 available values (3.81 (USNLM 2009); 3.69 (MRID 42970810); 3.8 (MRID 40226101)).

### Terrestrial plants (seeds, fruit, nectar and leaves)

Many listed species of birds consume plant matter, including seeds, fruit, nectar and leaves. T-REX EECs for these food items are depicted in **Figures 3-10 to 3-12**. Among these food items, residues on short grass are the highest, followed by broadleaves, tall grass and then seeds and fruit. Since insufficient data are available for estimating pesticide residues in nectar, the tall grass EEC is used as a surrogate for this food item. This is based on an analysis completed for the risk assessment methodology for honey bees[[26]](#footnote-27).

For seeds and fruit located on the treated field, upper bound peak EECs range 7.5-60 mg a.i./kg-food. Mean values range 3.5-28 mg a.i./kg-food. **Figure 3‑10** depicts the estimated concentrations of diazinon on seeds and fruit over time. When diazinon is applied at a single application of 0.5 lb a.i./A, diazinon upper bound residues are <0.01 mg a.i./kg-food 45 days after the application. For the maximum use scenario (cranberries), upper bound residues of diazinon reach <0.01 mg a.i./kg-food at 94 days after the first application.

Figure 3-10. Mean and upper bound estimated concentrations of diazinon on seeds and fruit.

For broadleaf plants located on the treated field, upper bound peak EECs range 69-540 mg a.i./kg-food. Mean values range 23-180 mg a.i./kg-food. **Figure 3-11** depicts the estimated concentrations of diazinon on broadleaf plants over time. When diazinon is applied at a single application of 0.5 lb a.i./A, diazinon upper bound residues are <0.01 mg a.i./kg-food 68 days after the application. For the maximum use scenario (cranberries), upper bound residues of diazinon reach <0.01 mg a.i./kg-food at 111 days after the first application.

Figure 3-11. Mean and upper bound estimated concentrations of diazinon on broadleaves.

For grass located on the treated field, upper bound peak EECs range 55-960 mg a.i./kg-food. Mean values range 18-340 mg a.i./kg-food. **Figure 3-12** depicts the estimated concentrations of diazinon on grass over time. When diazinon is applied at a single application of 0.5 lb a.i./A, diazinon upper bound residues are <0.01 mg a.i./kg-food at 66 and 72 days after the application for tall and short grass, respectively. For the maximum use scenario (cranberries), upper bound residues of diazinon reach <0.01 mg a.i./kg-food at 109 and 115 days after the application for tall and short grass, respectively.

Figure 3-12. Mean and upper bound estimated concentrations of diazinon on grass. Note that tall grass EECs are used as a surrogate for nectar.

### Terrestrial vertebrates (birds, mammals, amphibians, reptiles)

Diazinon concentrations in terrestrial vertebrate prey consuming grass or insects from treated areas are presented in **Table 3-24**. These estimates represent the peak values from mean and upper bound residues on food items directly sprayed with diazinon. As diazinon residues on grass and insects dissipate, residues would be expected to decrease in terrestrial vertebrate prey (*e.g.*, comparable to **Figures 3-10 to 3-12**). In addition, diazinon residues would likely be metabolized by terrestrial vertebrates to the non-toxic degradate, oxypyrimidine. Therefore, EECs in **Table 3-24** represent conservative estimates of diazinon concentrations in vertebrate prey.

The estimated concentrations of diazinon in terrestrial vertebrates are also used to represent concentrations in carrion. It is possible that exposed animals may die due to diazinon or other factors. For birds, some of the EECs overlap with levels where mortality is expected (LD50 values range 1.18-602 mg a.i./kg-bw). For mammals, the rat LD50 is 936 mg a.i./kg-bw (MRID 41334607).

## Estimated concentrations in aquatic food items (mg a.i./kg-food)

### Aquatic plants

No empirical bioconcentration factor (BCF) values are available for aquatic plants exposed to diazinon, and no data are available to describe the metabolism of diazinon by aquatic plants. Therefore, the KABAM generated BCF for phytoplankton, 280, is used to estimate diazinon concentrations in algae and aquatic plants that could potentially be consumed by listed birds (**Figure 3-13**). Based on this BCF, at an aqueous concentration of 1 µg a.i./L (ppb), the diazinon concentration in aquatic plants is 0.28 mg a.i./kg(food). At 1 mg a.i./L (ppm), the diazinon concentration in aquatic plants is 280 mg a.i./kg(food).

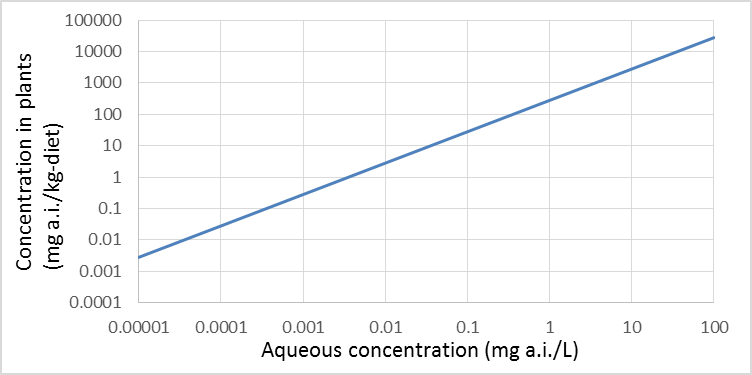


Figure 3-13. Diazinon concentrations in aquatic plants resulting from bioconcentration different aqueous concentrations.

### Aquatic invertebrates

Because a reliable metabolism rate constant cannot be generated to parameterize KABAM, the empirical BCF values for aquatic invertebrates and fish are used to estimate diazinon concentrations in aquatic organisms. Empirical BCFs for diazinon range 3-82 in aquatic invertebrates. EECs in aquatic habitats range from the parts per trillion to the parts per million range. The estimated concentrations in aquatic organisms resulting from this range of EECs are used in combination with the lower and upper bound of BCFs to bracket the potential concentrations of diazinon in aquatic invertebrate tissues (at steady state). **Figure 3-14** depicts the lower (blue) and upper (red) bounds. It should be noted that tissue concentrations in aquatic invertebrates will likely be bound by toxicity of diazinon on these organisms. For instance, the HC50 for aquatic invertebrates exposed to diazinon is 10 µg a.i./L. At this concentration, ≥50% mortality would be expected in 50 percent of the species. At 10 µg a.i./L, diazinon concentrations in aquatic invertebrate tissues are estimated at 0.030-0.82 mg a.i./kg(diet).

HC50

Figure 3-14. Upper (red) and lower (blue) of diazinon concentrations in aquatic invertebrates resulting from bioconcentration at different aqueous concentrations.

### Fish

Empirical BCFs for diazinon range 18-213 in fish. **Figure 3-15** depicts the lower (blue) and upper (red) bounds of diazinon concentrations in fish tissues resulting from environmentally relevant aqueous concentrations. Although fish are less sensitive to diazinon exposures compared to aquatic invertebrates, mortality to fish may also be a limitation of how much diazinon may be bioconcentrated in fish. At the fish HC50 of 3.4 mg a.i./L, diazinon concentrations in fish tissues are estimated at 61-720 mg a.i./kg(diet).

Figure 3-15. Upper (red) and lower (blue) of diazinon concentrations in fish resulting from bioconcentration at different aqueous concentrations.

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1. Based on the Toxic Release Inventory classification system where half-lives greater than 60 days in water, soil, and sediment are considered persistent and half-life greater than 6 months are considered very persistent (USEPA, 2012a). [↑](#footnote-ref-2)
2. Representative half-life values are generated for use as model inputs using the North American Free Trade Agreement (NAFTA) guidance for calculating degradation kinetics (NAFTA, 2012; USEPA, 2012c). The representative half-life may reflect both initial and later (potentially slower) portions of the decline curve and is not necessarily numerically similar to the value of the DT50, rather it provides a half-life input value for use in modeling that is generated using a standardized procedure from decay data that do not necessarily exhibit first-order behavior. The actual DT50 and DT90 from the representative degradation kinetic equations for the curve are used for descriptive purposes and for understanding the decline curve and the nature of the representative half-life used in modeling, see **APPENDIX 3-1 Environmental Transport and Fate Data Analysis** for these values. [↑](#footnote-ref-3)
3. Limited data are available for aerobic soil metabolism and one study had 30% of unaccounted for radioactivity in one soil. Because the study provided information on three soils and diazoxon, data were evaluated for radioactivity that was accounted for and radioactivity that was lost and could still be diazinon. The values calculated considering residues of diazinon alone and diazinon plus lost radioactivity are within the range of values measured in two other soils with acceptable mass balances. [↑](#footnote-ref-4)
4. Mobility was classified using the Food and Agriculture Organization (FAO) classification system (FAO, 2000) and supplemental sorption coefficients. [↑](#footnote-ref-5)
5. Diazoxon is a residue of concern for humans and other terrestrial and aquatic animals. [↑](#footnote-ref-6)
6. The current registered use pattern on apples allows for application of diazinon at 2 lbs a.i./A with one application during the dormant season and one application during the foliar season. [↑](#footnote-ref-7)
7. The ponds were immediately adjacent to the orchards but were not surrounded by orchard. [↑](#footnote-ref-8)
8. Average of individual samples collected from three different zones of the pond on the same day [↑](#footnote-ref-9)
9. There are currently no registered granular formulations in the United States. [↑](#footnote-ref-10)
10. The current registered application rate for use on cranberries is 3 lbs a.i/A with up to three applications. [↑](#footnote-ref-11)
11. The exposure models can be found at: <http://www.epa.gov/pesticides/science/models_pg.htm> [↑](#footnote-ref-12)
12. ‘NASS-HUC2’ worksheet in **APPENDIX 3-4a Diazinon Aquatic Modeling Inputs**. [↑](#footnote-ref-13)
13. http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-selecting-input-parameters-modeling (accessed November 13, 2015) [↑](#footnote-ref-14)
14. <http://www2.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-calculate-representative-half-life-values> (accessed November 13, 2015) [↑](#footnote-ref-15)
15. [http://www.regulations.gov/#!docketDetail;D=EPA-HQ-OPP-2013-0676](http://www.regulations.gov/) (accessed April 11, 2014) [↑](#footnote-ref-16)
16. Application dates were chosen to occur in the wettest month based on the meteorological file specific to each HUC2 regions. The application dates are listed in the **APPENDIX 3-4a Diazinon Aquatic Modeling Inputs.** [↑](#footnote-ref-17)
17. An aerial application was simulated with a buffer; however, the main simulation for ornamentals, did not simulate an aerial application without a buffer because aerial applications are not allowed for ornamentals. This explains how some EECs simulated with a buffer are slightly higher than the main simulation. Aerial applications are only allowed for applications to lettuce. [↑](#footnote-ref-18)
18. This analysis involved over 63,000 model simulations. [↑](#footnote-ref-19)
19. There are currently no registered granular formulations in the United States. [↑](#footnote-ref-20)
20. <http://www.epa.gov/pesticides/reregistration/diazinon/> [↑](#footnote-ref-21)
21. Specific state causes of impairment that make up the national pesticides cause of impairment group are listed at http://iaspub.epa.gov/tmdl\_waters10/attains\_nation\_cy.cause\_detail\_303d?p\_cause\_group\_id=885. [↑](#footnote-ref-22)
22. Specific state pollutants that make up the National Pesticides Pollutant Group and have TMDLs are listed at http://iaspub.epa.gov/tmdl\_waters10/attains\_nation.tmdl\_pollutant\_detail?p\_pollutant\_group\_id=885&p\_pollutant\_group\_name=PESTICIDES. [↑](#footnote-ref-23)
23. <http://water.epa.gov/action/advisories/drinking/upload/dwstandards2012.pdf> [↑](#footnote-ref-24)
24. California, Georgia, Virginia, Oregon, Utah, Texas, Indiana, Tennessee, Alabama, Wisconsin, and Louisiana [↑](#footnote-ref-25)
25. RED mitigations include cancellation of residential uses, seed treatments, and use of granules. Additionally, most aerial applications were cancelled. While these mitigations were implemented in prior to 2008, it may have taken some time for all products to be removed from the market. [↑](#footnote-ref-26)
26. USEPA. 2012. White Paper in Support of the Proposed Risk Assessment Process for Bees. Submitted to the FIFRA Scientific Advisory Panel for Review and Comment September 11 – 14, 2012. Office of Chemical Safety and Pollution Prevention Office of Pesticide Programs Environmental Fate and Effects Division, Environmental Protection Agency, Washington DC; Environmental Assessment Directorate, Pest Management Regulatory Agency, Health Canada, Ottawa, CN; California Department of Pesticide Regulation <http://www.cdpr.ca.gov/docs/emon/surfwtr/presentations/epa_whitepaper.pdf> [↑](#footnote-ref-27)