**Effects Determinations for Diazinon**

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# Introduction

For 1835 listed species, including endangered, threatened, candidate and proposed species, and 794 designated critical habitats, a “No Effect” (NE), “Not Likely to Adversely Affect” (NLAA) or a “Likely to Adversely Affect” (LAA) determination is made. For each species and designated critical habitat, the effects determination is based on the methodology previously described in **Section 1.4** of the Problem Formulation or a more qualitative analysis, depending on the listed species’ life history, unique habitat attributes (*e.g.,* deep ocean habitat), and/or the chemical use pattern overlap. These determinations are described further below.

# Summary of Effects Determinations

**Table 4-1** below summarizes the effects determinations for all species and designated critical habitats including a count of the number of species by taxon in each effects determination category. Effects determinations are summarized for each individual species and critical habitat in **APPENDIX 4-1** (this table is provided in Excel format due to its large size). **APPENDIX 4-1 (‘Summary Table All Calls’ tab)** is organized into 8 major taxa: birds, mammals, reptiles, amphibians, terrestrial invertebrates, fish, aquatic invertebrates and plants. Species are listed by taxa, then alphabetically according to scientific name, then by species identification number. For each species, the table includes an effects determination for both the species and their critical habitat, if applicable, as well as an indication of how the effects determination was reached (*e.g.,* terrestrial weight of evidence analysis, qualitative analysis – sea turtles, etc.). Each group of effects determinations is further described below.

**TABLE. 4-1. Summary of Species Effects Determinations for Diazinon (Counts by Taxon).**

|  |  |  |  |
| --- | --- | --- | --- |
| **TAXON** | **STEP 1 EFFECTS DETERMINATION** | **STEP 2 EFFECTS DETERMINATIONS** | **Totals** |
| **NO EFFECT** | **MAY AFFECT** | **NOT LIKELY TO ADVERSLY AFFECT** | **LIKELY TO ADVERSELY AFFECT** |
| Amphibians |  0 | 40 | 2 | 38 | 40 |
| Aquatic Invertebrates | 5 | 215 | 7 | 208 | 220 |
| Birds | 7 | 101 | 19 | 82 | 108 |
| Fish | 0 | 193 | 23 | 170 | 193 |
| Mammals | 3 | 106 | 25 | 81 | 109 |
| Plants | 69 | 892 | 199 | 693 | 961 |
| Reptiles | 1 | 47 | 0  | 47 | 48 |
| Terrestrial Invertebrates | 29 | 126 | 9 | 118 | 156 |
| Total | 114 | 1720 | 284 | 1437 | 1835 |
| Percentage of total # | 6% | 94% | 15% | 78% |  |

**TABLE. 4-2. Summary of Critical Habitat Effects Determinations for Diazinon (Counts by Taxon).**

|  |  |  |  |
| --- | --- | --- | --- |
| **DESIGNATED CRITICAL HABITAT TAXON** | **STEP 1 EFFECTS DETERMINATION** | **STEP 2 EFFECTS DETERMINATIONS** | **Totals** |
| **NO EFFECT** | **MAY AFFECT** | **NOT LIKELY TO ADVERSLY AFFECT** | **LIKELY TO ADVERSELY AFFECT** |
| Amphibians | 2 | 23 | 2 | 20 | 25 |
| Aquatic Invertebrates | 3 | 72 | 7 | 65 | 75 |
| Birds | 5 | 26 | 4 | 22 | 31 |
| Fish | 0 | 106 | 9 | 97 | 106 |
| Mammals | 2 | 30 | 8 | 23 | 32 |
| Plants | 59 | 403 | 280 | 123 | 462 |
| Reptiles | 2 | 15 | 5 | 10 | 17 |
| Terrestrial Invertebrates | 10 | 36 | 11 | 25 | 46 |
| Total | 83 | 711 | 326 | 385 | 794 |
| Percentages of Total number | 10% | 90% | 41% | 48% |  |

# Effects Determinations of No Effect

The Step 1 “No Effect/May Affect” analysis determines if a listed species or its designated critical habitat requires further analysis due to the potential for co-occurrence with the action area. A critical habitat determination may differ from the species determination when the critical habitat extends outside of the species range. The spatial footprint of the action area includes the pesticide footprint based on all labeled uses for the chemical and offsite transport due to both spray drift and downstream dilution. Additional information on how the action area was developed can be found in **Attachment 1-3**, and additional information on the downstream dilution analysis can be found in **Appendix 3-5**. The species’ range geospatial files used in this analysis were provided by the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS).

When no co-occurrence is identified between the listed species range and the action area, the species and its designated critical habitat receives a “No Effect” determination. A “May Affect” determination is given if co-occurrence exists, moving the species and its designated critical habitat to Step 2 for further analysis. Co-occurrence was determined by overlaying the action area and with the species range and designated critical habitat using ArcGIS v10.4.

“No Effect” determinations were made for species with no designated critical habitat and meeting at least one of the following criteria: a) the species is presumed by the U.S. Fish and Wildlife Service (USFWS) to be extinct; b) the species no longer occurs in the US; or c) the species exists only in captivity. Species categorized as “presumed extinct” are often difficult to ascertain based on the frequent updates to information. It is possible that the effects determination for species in this category may change if new or different information becomes available.

“No Effect” determinations are made due to presumed extinction for 16 species including 5 species of birds, 2 species of mammals and 9 species of terrestrial invertebrates (**APPENDIX 4-1 (‘NE\_Extinct’ tab)**).

Of the remaining 1819 listed species not presumed to be extinct, 1721 overlap with the diazinon action area, and receive a “May Affect” determination and further consideration in Step 2. “No Effect” determinations are made for 98 species including 2 bird species, 2 crustacean species, 1 coral species, 3 ferns and allies species, 66 flowering plant species, 10 insects species, 1 mammal, 1 reptile, and 12 snail species (**APPENDIX 4-1 (‘NE\_Diaz’ tab))**.

Of the 794 designated critical habitat locations, 711 overlap with the diazinon action area, and receive a “May Affect” determination and further assessment in Step 2. “No Effect” determinations are made for 83 species including 2 amphibian species, 2 arachnid species, 5 bird species, 2 coral species, 2 ferns and allies species, 57 flowering plants species, 8 insect species, 2 mammal species, 2 reptile species and 1 snail species (**APPENDIX 4-1 (‘NE\_Diaz’ tab))**.

# Effects Determinations of NLAA - No overlap

The Step 2 overlap analysis uses the results from the Step 1 analysis to calculate the percent overlap of the species range or its designated critical habitat with each use site included in the action area; see **Attachment 1-6** for results of the Step 1 and Step 2 analysis. Additional information on how the action area and use sites were developed can be found in **Attachment 1-3**.

Species of interest were identified by determining percent of the ranges and/or designated critical habitats of listed species impacted directly by the use sites and the percent impacted by spray drift. In order to calculate the percent overlap, first the total acres of use within the species range was calculated from the total number of co-occurring use site raster cells. To calculate the percent overlap, the total acres of use was divided by the total acres of the species file occurring within the spatial extent of the use site.

One known source of error within spatial datasets is positional accuracy and precision. The National Standard for Spatial Data Accuracy outlines the accepted method for calculating the horizontal accuracy of a spatial dataset (FGDC 1998)[[1]](#footnote-1). To prevent false precision when calculating area and the percent overlap, only two significant digits should be considered for decision purposes given the reported 60 meters of horizontal accuracy for the Cropland Data Layer (CDL).

An effects determination of “Not Likely to Adversely Affect” is reached for 13 species that were “presumed extinct” based on information gathered in review of the 5 year status review but did not meet the additional criteria to receive a no effect determination. “Not Likely to Adversely Affect” determinations were reached for 1 amphibian species, 8 species of birds, 3 species of mammals and 1 terrestrial invertebrate species (**APPENDIX 4-1 (‘NLAA\_Extinct’ tab)**). It is possible that the effect determination for species in this category may change if new information becomes available.  Critical habitat for “presumed extinct” species was assessed further before making a determination specific to the critical habitat.

“Not Likely to Adversely Affect” determinations are made for species and/or critical habitats occurring exclusively on the uninhabited Northwestern Hawaiian Islands of Nihoa and Laysan Island.  Based on the 2000 U.S Census, the human population of these islands is zero[[2]](#footnote-2).  As a result, human presence on these islands is expected to be zero to extremely low and the potential exposure to chlorpyrifos is considered to be discountable. Consequently, ​​“Not Likely to Adversely Affect” determinations were made for 4 birds species, and 2 flowering plants species found only on these islands and their associated critical habitat,​and for the critical habitat for an additional flowering plant species (**APPENDIX 4-1 (‘NLAA\_OutsideUse’ tab)**).

Midway Islands, the pacific island coral atolls and Mona Island of Puerto Rico have low human populations ranging from 7-150 people, based on the 2000 U.S Census2. While human impact is expected to be low on these islands, the possibility of exposure is not discountable and the 4 species and/or critical habitats occurring exclusively on these islands are considered further.

Other Minor Outlaying U.S Islands such as, Wake Island, the additional Northwestern Hawaiian Islands and minor islands of Hawaii were also considered. However, none of these species occur exclusively in these locations.  Therefore, exposure was not discountable based on location alone**,** and these species were considered further before making a determination.

A “Not Likely to Adversely Affect” determination is made for those species and/or designated critical habitats for which the use site (including off-site transport) and range overlap is less than 1% after rounding for significant digits. All species with 1% or greater overlap were assessed further before assigning an effects determination. A critical habitat determination may differ from the species determination when the critical habitat extends outside of the species range or the location and size increases the impact due to spray to 1% or greater.

Species inhabiting flowing aquatic habitat were assessed in the downstream dilution analysis and excluded from this analysis; additional information on the downstream dilution analysis can be found in Appendix 3-5. Species that received a “Not Likely to Adversely Affect” based on the downstream dilution analysis can found in **APPENDIX 4-1(‘NLAA\_Diaz\_DD’ tab)**.

Of the remaining 1701 species where a ‘May Affect’ determination was reached for diazinon in Step 1, 1591 species were assessed further before assigning an effects determination. The remaining 110 were impacted by spray drift only in less than 1% of their range and received a “Not Likely to Adversely Affect” determination **APPENDIX 4-1 (‘NLAA\_Diaz\_Overlap’ tab).**

“Not Likely to Adversely Affect” determinations were made for 1 amphibian species, 2 bird species, 7 ferns and allies species, 4 fish species, 84 flowering plants species, 3 insect species, 3 mammal species, and 6 snail species, based on lack of spatial overlap between species ranges and diazinon use sites.

Of the remaining 707 ‘May Affect’ designated critical habitat determinations for diazinon, 517 were assessed further before assigning an effect determination. The remaining 190 were impacted by only spray drift in less than 1% of the designated critical habitat and received a “Not Likely to Adversely Affect” determination **APPENDIX 4-1 (‘NLAA\_Diaz\_Overlap’ tab).**

“Not Likely to Adversely Affect” determinations were made for 2 arachnid species, 4 bird species, 5 ferns and allies species, 2 fish species, 172 flowering plant species, 3 insect species, and 2 snail species.

Species that only co-occur within the cattle ear tag footprint were assessed in the cattle ear tag analysis and excluded from this analysis; additional information on the cattle ear tag analysis can be found in section 7. Sea turtles, whales and deep sea fish, marine mammals (excluding whales), cave dwelling invertebrate species, lichens and pinnipeds and otters, were assessed in separate analyses and excluded from this analysis; additional information on these analyses can be found in section 7.

# Effects Determinations of NLAA/LAA: Weight of Evidence Analysis

A weight of evidence analysis, as described in **Section 1.4.2.2** of the Problem Formulation, is used to make effects determinations on 1375 species. The weight of evidence analysis was completed by producing “matrices” that capture the multiple lines of evidence. Direct effects considered for listed animals included effects on mortality, growth, reproduction, behavior and sensory function and indirect effects considered included impacts to prey/dietary items, habitat and obligate organisms. For plants, lines of evidence considered included mortality, growth, and reproduction and indirect effects considered included impacts to pollinators, habitat and obligate organisms. Additional lines of evidence addressing direct and indirect effects due to chemical mixtures and stressors and effects due to abiotic stressors (see **ATTACHMENT 4-1**) were also considered if an effects determination could not be reached based on the direct and indirect lines of evidence alone (*i.e.,* if LAA determination was reached based on the direct/indirect lines, analysis of additional data was not conducted). Depending on the species primary habitat, a terrestrial or aquatic weight of evidence matrix was completed. Some species life history dictated the need for both a terrestrial and aquatic weight of evidence matrix to be created to fully characterize potential risk to the species. If the listed species spent limited time in a different habitat, a second matrix was not developed; rather, an indication of the additional aquatic or terrestrial exposure was included in the primary matrix (*e.g.,* the lower keys marsh rabbit, a terrestrial species which spends time in wetland/marsh environment). Information gathered from the species range, overlap with the chemical use patterns, dietary items, EECs for both terrestrial and aquatic exposures, and indirect relationships and effects are integrated into each species matrix to make the determination of risk and confidence in data with a corresponding high, medium or low rating. Criteria used to determine if a line of evidence warrants a high, medium or low rating are described in **ATTACHMENT 1-9**. For designated critical habitats, any potential for direct or indirect effects to a listed species, based on the lines of evidence, are considered for the effects determination, regardless if the listed species is present within and/or currently inhabits the designated critical habitat (as not all designated critical habitats are currently occupied by individuals of a listed species).

Summary results for the species determinations based on the weight of evidence matrices are contained in **APPENDIX 4-1 (‘Summary’ tab)** and are denoted as either “TerrWoE”, “AquaWoE” or “TerrWoE and AquaWoE” as the source of the effects determination. Additional worksheets in **APPENDIX 4-1** include summary information on individual lines of evidence for each species and a key to file locations for each species. Detailed matrices for all species are located in **APPENDIX 4-3**, organized by species taxa and order. A detailed discussion of the weight of evidence matrices, including how exceedances of thresholds were determined and detailed discussion of each risk line, is included in **ATTACHMENT 4-1**. Criteria used to determine if a line of evidence warranted a high, medium or low rating are described in **ATTACHMENT 1-9**.

# Effects Determinations of LAA: Downstream dilution

While exposure to pesticide use is evaluated near the range of the species taking into account use overlap, runoff, and spray drift from nearby fields, the impact of upstream contributions of pesticide use is assessed via the downstream dilution analysis.  This is of particular interest for species with ranges or critical habitats that do not overlap with land covers representing areas where diazinon may be applied via spray (*i.e.*, orchards, ground fruit and vegetable and nurseries).

EPA does not currently have an approved flowing water model to allow for the evaluation of downstream processes (*e.g.*, advection dispersion, turbulent mixing, aquatic degradation, etc.) over long stream distances.  Instead, EPA uses the percent use area for watershed catchments as a surrogate for potential pesticide contributions and potential sources of dilution.  More details of the approach and the results of the downstream dilution analysis are provided in **Appendix 3-5**.

Based on the analysis, three species are identified that do not overlap with orchard, ground fruit and vegetable and nursery land covers; however, the downstream dilution analysis indicates that upstream applications may lead to concentrations that are sufficient to exceed direct and/or indirect effects thresholds. Therefore, LAA determinations are made for these three species, which include:

* Hay’s spring amphipod (*Stygobromus hayi*; entity ID 475);
* Little Aguja Pondweed (*Potamogeton clystocarpus*; Entity ID 807); and
* Short-tailed albatross (*Phoebastria albatrus*; Entity ID 88).

# Effects Determinations of NLAA/LAA: Qualitative Analyses

Not all listed species were included in the WoE effects determinations presented in Section 5.  Section 7 contains the effect determinations for these listed species based on qualitative analyses. Potential risk to these listed species were assessed qualitatively because EPA does not currently have methods available to adequately estimate potential exposures for these species. In many cases, these species live exclusively (*i.e.,* whales, deep fish) or primarily (*i.e.,* sea turtles, marine mammals) in marine environments, or are cave dwellers (invertebrate species). Other qualitative analyses focus on certain uses (*i.e.*, cattle ear tag use, seed and granular treatment) for which reliable exposure methods are not available as current terrestrial methods are suited for foliar [flowable] applications.

## Sea Turtle Analysis

This assessment considers the effects of diazinon on 9 listed species of sea turtles, including 2 listings of the loggerhead (different DPSs), 6 listings of the green, the leatherback, the hawksbill, kemp’s ridley and 2 listings of the olive ridley sea turtle. The biological information (*e.g.,* diet, habitat) used in this assessment for each species is included in **Attachment 1-14**.

Sea turtles spend the vast majority of their lives in aquatic habitats. Hatchling, juvenile and adult sea turtles forage in offshore and nearshore coastal habitats, including estuaries[[3]](#footnote-3). Adult sea turtles use ocean habitats during the majority of their lives. In addition, green sea turtles use freshwater streams and rivers during part of their lifecycle. None of the other listed sea turtles use freshwater habitats. While in aquatic habitats, exposure to diazinon via contaminated dietary items is assessed. Other exposure routes (*e.g.,* dermal) are not assessed in aquatic habitats because methods are not available to estimate exposures via non-dietary routes. Other routes of exposure are assessed in terrestrial habitats, *i.e.,* beaches. Sea turtles utilize beaches to lay their eggs, while some species use beaches to bask. As a result, eggs, hatchlings and adults may be exposed to diazinon from spray drift transport from treatment sites that are adjacent to nesting sites. Exposure routes of concern include inhalation and dermal interception of spray droplets on the day of the application. Since sea turtles do not forage while on land, dietary exposure while in terrestrial habitats is not expected.

This section provides the risk assessment used to assess exposures and potential effects to sea turtles on beaches and in their aquatic habitats. Indirect effects to prey and habitat are also assessed in aquatic habitats. Effects determinations for the species and their designated critical habitats are based on the risk assessments that integrate the weight of evidence based on exposure, available effects data and the uncertainties associated with these data.

**7.1.1. Risk Assessment for Aquatic Habitats: Direct Effects**

*Dietary toxicity data*

As indicated above, dietary exposure is assessed for aquatic habitats. No dietary toxicity data are available for reptiles exposed to diazinon. As a result, toxicity data available for birds are used as a surrogate for reptiles. For dietary exposure, the avian thresholds for mortality and sublethal effects are 2.5 and 4.0 mg a.i./kg-diet, respectively (see **Table 6-1** of **Chapter 2**). Other endpoints considered in this analysis include the lowest avian LC50 (*i.e.,* 38 mg a.i./kg-diet) and the NOEC and LOEC where effects to avian reproduction were observed (*i.e.,* 8.3 and 16.3 mg a.i./kg-diet, respectively).

When considering bioaccumulation in aquatic organisms, pesticide uptake occurs through two routes: respiration of water and consumption (dietary) of contaminated food items. Bioaccumulation is limited by the extent to which a pesticide is eliminated (through respiration or fecal elimination) or metabolized to non-toxic degradates. For chemicals with Log Kow < 4, exposure from food becomes insignificant because uptake and depuration through respiration controls the residue in the organism. For chemicals with Log Kow > 6, diet is the predominant uptake route, with little influence from respiration. Given that the Log Kow value of diazinon is 3.77, Bioconcentration Factors (BCFs), which are based on studies involving respiratory uptake only, are considered representative of bioaccumulation because the dietary route is not substantial relative to respiration. Diazinon is not expected to accumulate over time in prey because it is not persistent in aqueous environments (Chapter 3), and is readily excreted and metabolized by animals[[4]](#footnote-4),[[5]](#footnote-5). BCFs for diazinon in aquatic plants, invertebrates and fish are used predict short term uptake of diazinon in aquatic organisms that represent the prey of listed sea turtles.

Bioconcentration factors (BCFs) for diazinon for the three relevant food items of sea turtles (*i.e.,* aquatic plants, invertebrates and fish), are used to translate toxicity thresholds to aquatic concentrations that would constitute a potential risk from dietary exposure. In this approach, the threshold is divided by the BCF. Since a BCF represents the ratio of the chemical concentration in animal tissue (mg a.i./kg-diet) to the concentration in water (mg a.i./L-water), this approach assumes that the threshold is equivalent to the chemical concentration in animal tissue.

As noted in **Chapter 3** (exposure characterization), the KABAM[[6]](#footnote-6)-estimated BCF for plants is 280. This value is uncertain because it is based on a model estimate that does not account for metabolism of diazinon by plants. For aquatic invertebrates and fish, several empirically based values are available from different studies (7 invertebrate species, 10 fish species), leading to increased confidence in available values. Minimum, maximum and mean of empirical BCF values for aquatic invertebrates and fish are provided in **Table 4-3**. These BCFs were used to estimate aqueous concentrations sufficient to exceed the thresholds and endpoints where effects were observed (*i.e.,* lowest LC50 and reproduction LOEC). Given that these BCF values are based on steady state, it is necessary to compare EECs that are representative of the time to steady state for diazinon. Based on the Log Kow of diazinon (3.7), steady state is reached in approximately 4 days. Therefore, in this assessment, 4-day average water concentrations are compared to the values in **Table 4-3**.

**Table 4-3. Aqueous concentrations defining dietary exposures of concern for sea turtles (all ages).**

|  |  |  |  |
| --- | --- | --- | --- |
| **Food item** | **Description of BCF** | **BCF value** | **Aqueous concentration (µg a.i./L) above which there is concern for dietary exposure** |
| **Mortality threshold** | **Lowest LC50** | **Sublethal threshold\*** | **Lowest NOEC for repro** | **Lowest LOEC for repro** |
| Aquatic plants | KABAM estimate | 280 | 9 | 136 | 14 | 30 | 58 |
| Aquatic invertebrates | lowest of empirical | 3 | 833 | 12667 | 1333 | 2767 | 5433 |
| mean of empirical | 25 | 100 | 1520 | 160 | 332 | 652 |
| highest of empirical | 82 | 30 | 463 | 49 | 101 | 199 |
| Fish | lowest of empirical | 18 | 139 | 2111 | 222 | 461 | 906 |
| mean of empirical | 88 | 28 | 432 | 45 | 94 | 185 |
| highest of empirical | 213 | 12 | 178 | 19 | 39 | 77 |

*\*Based on AChE inhibition*

There is considerable uncertainty in using birds as surrogates for reptiles as it is assumed that they will have similar responses to diazinon. The actual sensitivities of reptiles to diazinon relative to birds is unknown. Given that diazinon’s toxicity is attributed to metabolic transformation to the oxon degradate, differences in metabolic rates of birds and reptiles may lead to different sensitivities. Since birds are warm blooded and reptiles are cold blooded, it is expected that the metabolic rates of reptiles will be lower than those of birds. Limited data are available to examine the differences in toxicity of birds and cold-blooded species. Available toxicity data for one cold-blooded species, the bullfrog, is at least 3 orders of magnitude less sensitive to diazinon compared to birds tested in the same study (mallards and pheasants; ECOTOX 50396[[7]](#footnote-7)). Therefore, there is low confidence associated with the robustness and relevance of use of the available avian toxicity data as a surrogate for reptiles.

*Exposures in estuaries and near shore areas*

There is a great deal of uncertainty in estimating potential diazinon exposures in estuaries and near-shore areas of the ocean because the existing fate and transport models do not account for water stratification, tidal flux, and complex currents that occur in these habitats (*i.e.*, bins 8 and 9). As noted in the problem formulation (**Chapter 1**), estimates developed for aquatic bin 2 are generally used as surrogate exposure levels for intertidal nearshore waterbodies (bin 8), and estimates developed using aquatic bin 3 are generally used as surrogate exposure levels for subtidal nearshore waterbodies (bin 9). Additionally, aquatic bin 5 is generally used as a surrogate for tidal pools occurring during low tide (aquatic bin 8). The 4-day average EECs for bins 2, 3, and 5, which include both runoff and drift, are 186-2040, 6.23-315, and 236-4250 µg a.i./L, respectively (EECs are presented in **Appendix 3-4b**; values based on HUCs 1-3, 8, 12, 13, 17-21, which are relevant to the species ranges). These 1-in-15 year 4-day average EECs exceed several of the concentrations that represent threshold and endpoint exceedances listed in **Table 4-3**. Therefore, there is concern for potential direct effects to sea turtles exposed via diet while in estuaries and near shore habitats.

It should be noted that there is a great deal of uncertainty surrounding the EECs for listed sea turtles. The marine bins selected for these species (bins 8 and 9) cannot be modeled, so surrogate freshwater bins (bins 2, 3, and 5) are used. Bins 2 and 3 are used to represent low and high tide periods, with bin 5 representing a tidal pool. The EECs for these bins reflect contributions from both runoff and spray drift from treated areas adjacent to the waterbodies, which may not typically occur for intertidal and subtidal nearshore waterbodies which have beaches between the treated area and the leading edge of the water. Fate data (*e.g.*, hydrolysis, metabolism) are not available for diazinon in saltwater environments, so it is uncertain how representative the EECs are for a marine environment. While the flowing bins (bins 2 and 3) are being used to represent the exchange of water expected in the marine bins due to the tides, there is uncertainty in how well these bins reflect the turbulent, mixing nature of the 12-hour tidal cycle and the EECs that may be present. The surrogate bins have lower depths and widths than the bins they are designed to represent: a 0.1 m depth and 1-2 m width for bins 2 and 5 compared to 0.5 m depth and 50 m width for bin 8 and a 1 m depth and 8 m width for bin 3 compared to 5 m depth and 50 m width for bin 9. While the smaller bins (bins 2 and 5) could be used for rearing juveniles, it is uncertain if these surrogate bins could hold sufficient volume to contain adult sea turtles. The additional depth and width assigned to bins 8 and 9 could also result in lower EECs based on the additional water volume and hence dilution.

Some monitoring data are available where diazinon concentrations are measured in estuaries. Concentrations reported in CA in 2008-2009 are <1µg a.i./L[[8]](#footnote-8). It should be noted that the utility of these data are limited in that they represent ambient monitoring data for which the applications of diazinon in the watershed of the sampled estuaries are not defined and are not expected to capture peak exposures.

EECs resulting from drift only transport into estuaries are provided in **Tables 4-4** (maximum ground spray applications of 4 lb a.i./A) **and 4-5** (maximum aerial application of 2 lb a.i./A). For bins 2 and 5 located within several hundred meters of treatment sites, EECs are at levels that pose a risk to turtles.

**Table 4-4. Aquatic EECs (µg a.i./L) resulting from spray drift from ground application at 4 lb a.i./A (maximum rate allowed). Distances represent different distances between application site and water body. EECs are instantaneous values (not 1-in-15 year 4-d average EECs).**

|  |  |
| --- | --- |
| **Distance (m)** | **EEC by Bin** |
| **2** | **3** | **4** | **5** |
| 0 | 3200 | 150 | 22 | 3661 |
| 30 | 113 | 10 | 3.3 | 115 |
| 60 | 51 | 4.8 | 1.8 | 51 |
| 90 | 31 | 3.0 | 1.2 | 31 |
| 120 | 22 | 2.1 | 0.88 | 22 |
| 150 | 17 | 1.6 | 0.69 | 17 |
| 180 | 13 | 1.3 | 0.57 | 13 |
| 210 | 11 | 1.1 | 0.48 | 11 |
| 240 | 9.5 | 0.93 | 0.41 | 9.5 |
| 270 | 8.2 | 0.81 | 0.36 | 8.2 |
| 300 | 7.2 | 0.71 | 0.32 | 7.2 |
| 304 | 7.1 | 0.70 | 0.31 | 7.1 |

**Table 4-5. Aquatic EECs (µg a.i./L) resulting from spray drift from aerial application at 2 lb a.i./A (maximum rate allowed). Distances represent different distances between application site and water body. EECs are instantaneous values (not 1-in-15 year 4-d average EECs).**

|  |  |
| --- | --- |
| **Distance (m)** | **EEC by Bin** |
| **2** | **3** | **4** | **5** |
| 0 | 994 | 75 | 17 | 1050 |
| 30 | 198 | 18 | 6 | 201 |
| 60 | 109 | 10 | 4 | 110 |
| 90 | 75 | 7.3 | 3.0 | 75 |
| 120 | 57 | 5.6 | 2.4 | 57 |
| 150 | 46 | 4.5 | 1.9 | 46 |
| 180 | 38 | 3.8 | 1.7 | 38 |
| 210 | 33 | 3.2 | 1.4 | 33 |
| 240 | 29 | 2.8 | 1.3 | 29 |
| 270 | 26 | 2.5 | 1.1 | 26 |
| 300 | 23 | 2.3 | 1.0 | 23 |
| 304 | 23 | 2.3 | 1.0 | 23 |
| 330 | 21 | 2.1 | 0.94 | 21 |
| 360 | 19 | 1.9 | 0.86 | 19 |
| 390 | 18 | 1.8 | 0.80 | 18 |
| 420 | 16 | 1.6 | 0.74 | 16 |
| 450 | 15 | 1.5 | 0.70 | 15 |
| 480 | 14 | 1.4 | 0.65 | 14 |
| 510 | 13 | 1.3 | 0.62 | 14 |
| 540 | 13 | 1.3 | 0.58 | 13 |
| 570 | 12 | 1.2 | 0.55 | 12 |
| 600 | 11 | 1.1 | 0.53 | 11 |
| 630 | 11 | 1.1 | 0.50 | 11 |
| 660 | 10 | 1.0 | 0.48 | 10 |
| 690 | 9.9 | 0.99 | 0.46 | 9.9 |
| 720 | 9.5 | 0.95 | 0.44 | 9.5 |
| 750 | 9.1 | 0.91 | 0.42 | 9.1 |
| 780 | 1.1 | 0.11 | 0.05 | 1.1 |
| 793 | 1.1 | 0.11 | 0.05 | 1.1 |

*Exposures in Off-Shore Habitats*

It is expected that, given the large volume of water in oceans, and the lack of persistence of diazinon, this chemical will be sufficiently diluted to not be of concern for adult sea turtles in deep water ocean habitats. In addition, since diazinon is readily metabolized and does not accumulate in aquatic organisms, exposure via consumption of aquatic plants, invertebrates and fish is not of concern.

It should be noted that this approach differs from that taken above for smaller, near shore habitats (*e.g.,* estuaries). In these areas, short term concentration of diazinon in aquatic prey of sea turtles is considered because concentrations may be higher in water and diazinon could concentrate for a short term period in prey of sea turtles. The use of empirically based bioconcentration factors allows for consideration of metabolism and for uptake in prey (fish, invertebrates) via the major route of exposure (*i.e.,* respiration).

*Exposures in freshwater environments (for Green Sea Turtle only)*

Aquatic bins that are used as surrogates for estimating exposures to green sea turtles in freshwater habitats are bins 3 and 4 (Attachment 1-10). The 1-in-15 year 4-day average EECs, which include both runoff and drift, derived for these bins are on the order of 6.23-315 and 4.72-351 µg a.i./L, respectively (Bin 3 and 4 EECs are presented in **Appendix 3-4b**; values based on HUCs 1-3, 8, 12, 13, 17-21). These 1-in-15 year 4-day average EECs exceed several of the concentrations that represent threshold and endpoint exceedances listed in **Table 4-3**. Therefore, there is concern for potential direct effects to green sea turtles exposed via diet while in freshwater habitats.

**7.1.2. Risk Assessment for Aquatic Habitats: Indirect Effects**

**Table 4-6** summarizes the thresholds used to assess indirect effects to sea turtles through impacts to diet (aquatic plants, invertebrates and fish) or habitat (plants). Details of how these values were derived are provided in **Chapter 2**.

**Table 4-6. Thresholds for indirect effects (µg a.i./L).**

|  |  |  |
| --- | --- | --- |
| **Taxa** | **Mortality threshold** | **Sublethal threshold** |
| Aquatic plants | Not applicable | 3700 |
| Aquatic invertebrates | 0.259 | 0.42 |
| Fish | 123.5 | 0.47 |

For estuaries and near shore ocean habitats, 1-in-15 year average daily EECs estimated for bins 2, 3, and 5 are on the order of 15-6070 µg a.i./L (EECs for bins 2, 3, and 5 are presented in **Table 3-10**). These values are above thresholds for mortality and sublethal thresholds for aquatic plants, aquatic invertebrates and fish (**Table 4-6**).

For sea turtles in the off shore habitats, indirect effects due to loss of prey or habitat are not expected. This is due to the effect of dilution in deep water ocean environments in which the adult sea turtles are found.

For freshwater habitats used by the green sea turtle, 1-in-15 year average daily EECs estimated for bins 3 and 4 are on the order of 14-947 µg a.i./L. These values are above thresholds for mortality and sublethal thresholds for aquatic invertebrates and fish, suggesting indirect effects may occur due to possible reductions in these resources (**Table** **4-6**). EECs are below the threshold for aquatic plants, therefore, effects to plant dietary items and habitat are not expected.

**7.1.3. Risk Assessment for Terrestrial Habitats (*i.e.*, Beaches)**

Spray drift transport from use sites adjacent to beaches could potentially result in exposures to eggs, hatchlings (leaving the nest) and adults (laying eggs or basking).

*Exposures to eggs*

When considering exposures to eggs, no toxicity data are available where reptile eggs were exposed to diazinon. Two studies are available where bird eggs were directly treated with diazinon. In ECOTOX 40200, no effects to hatchability or immune response were observed in bobwhite quail eggs directly sprayed with 4.0 lb a.i./A of formulated diazinon, which is the maximum application rate allowed on labels. This suggests no effects to embryos. In the second study, (Hoffman and Eastin 1981; ECOTOX#35250) a 11% decrease was reported in weight of mallard embryos of eggs treated with 2 lb a.i./A diazinon in oil. When treated at 15 lb a.i./A in an aqueous emulsion, no effects to growth were observed. There is a great deal of uncertainty associated with the reported application rates because the extrapolation to field rate is unverifiable as no math was provided by the study authors.

Spray drift transport is of potential concern for exposure. If spray drift deposition is simulated from the maximum rate of 4 lb a.i./A (ground spray) to deposition of 2 lb a.i./A, the potential area of concern extends 7 feet from the edge of the field.

This is considered to be discountable because, 1) sea turtle nests are unlikely to be located near treated fields (*i.e.*, within 7 feet from the edge of the field) as these areas are likely to be disturbed, 2) it is expected that spray drift deposition will also be intercepted by sand that covers sea turtle nests, substantially decreasing the exposure of eggs to diazinon transported via spray drift; and 3) since eggs are laid most frequently at night and diazinon applications occur during the day, eggs are not expected to be uncovered at times when spray drift exposure occurs. Therefore, the available information suggests that diazinon does not pose a risk to sea turtle eggs.

This approach discussed above assumes that the 100% of the pesticide transported via spray drift infiltrates through the sand covering the nest. Pesticide transport via runoff is not considered here because methods are not available to translate concentrations in water to an exposure relevant to eggs.

*Dermal exposures to hatchlings and adults*

When considering exposures to hatchlings and adults on beaches, dermal exposure due to spray drift transport is considered to be of concern. Toxicity data are not available for dermal exposures involving reptiles. There are data available to suggest that this route may cause effects in birds and mammals. Effects observed in Canada goose included mortality and AChE inhibition at exposures of 0.5-4 lb a.i./A (Vyas *et al*. 2006 ECOTOX #85970); however, the exposure included both dietary and dermal exposures, so the relative influence of the dermal route cannot be quantified. The 1/million threshold (*i.e.*, 0.0032 lb a.i./A), LC10 (0.091 lb a.i./A) and LC50 (0.31 lb a.i./A) generated from the data presented in that study (see **Chapter 2**, section on bird toxicity data) are used here. Although these values are all based on mortality related effects, they also bracket levels where sublethal effects are observed in the same study. At 0.25 lb a.i./A, which is the lowest treatment level of the study, 14% inhibition of AChE is observed.

Deposition equivalent to the 1/million threshold of 0.0032 lb a.i./A extends hundreds of feet from the edge of the field for all application rates of diazinon for ground, aerial and air blast application methods. Deposition is equivalent to LC10 and LC50 values for up to 112 feet from the edge of the field for ground and air blast applications and up to 853 feet for aerial applications (**Table 4-7**). There is considerable uncertainty in using the data from the Vyas *et. al* study for the purpose of quantifying effects due to dermal exposure as these endpoints are due to a combination of dietary and dermal exposure. It is expected that there is no dietary exposure to hatchling and adult sea turtles while on beaches because they consume aquatic organisms (*i.e.*, aquatic plants, invertebrates and fish).

**Table 4-7. Distance from edge of field to reach deposition equivalent to 1/million threshold for Canada geese, LC10 and LC50 based on data from Vyas *et. al*.**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Application method** | **Rate** **(lb a.i./A)** | **Distance to 1/million threshold** **(0.0032 lb a.i./A)** | **Distance to LC10** **(0.091 lb a.i./A)** | **Distance to LC50 (0.31 lb a.i./A)** |
| Ground (high boom, very fine to fine spectrum) | 0.5 | 344 | 16 | 3 |
| 1 | 581 | 30 | 10 |
| 2 | 922 | 56 | 20 |
| 3 | >1000 | 82 | 26 |
| 4 | >1000 | 112 | 33 |
| Aerial (very fine to fine) | 1 | >1000 | 341 | 49 |
| 2 | >1000 | 853 | 174 |
| Air blast | 0.5 | 128 | 16 | 0 |
| 1 | 174 | 30 | 0 |
| 2 | 236 | 46 | 0 |

There is a high degree of uncertainty associated with concluding that dermal exposures of diazinon will pose a risk to sea turtles. First, in order for an exposure to occur, the application must occur either on the day when turtles are basking, when a female lays eggs or when the hatchlings leave the nest. Second, there is additional uncertainty associated with the likelihood of exposure because on the day of application, pesticide must be transported by wind blowing from the application site toward the beach. **Table 4-8** provides a summary of the percentage of time the wind is blowing from a certain direction. These data suggest that prevailing winds blow from inland toward the ocean up to 30% of the time (independent of when diazinon is likely to be applied). Third, the duration of potential exposures would be limited. For hatchlings, the potential for dermal exposures would only occur during their movement from their nesting site to the water – which occurs fairly rapidly (for predator avoidance). Once in the water, it is expected that any residues not immediately absorbed would be washed off. For females that are laying eggs, potential dermal exposures would occur for one night during their movement from the water to the nesting site, and then from the nesting site to the water. The number of clutches laid per year varies by species (*e.g.,* Loggerhead and green sea turtles nest 2-5 times per year, Leatherbacks will nest 7-10 times per year), so, there may be differences in the likelihood of exposure among species. Another important uncertainty to consider in the likelihood of effects is associated with the available toxicity data. As noted above, the relative sensitivities of reptiles and birds are unknown, leading to uncertainty associated with the representativeness of using the Canada goose effects data to represent potential mortality and AChE inhibition effects in sea turtles. In addition, the available data represent a combination of both dermal and dietary exposures; however, dietary exposures are not of concern for terrestrial habitats, leading to an overestimation of the toxicity of diazinon through the dermal route. Taken together, the chances of dermal exposure to diazinon and resulting effects are expected to be low.

**Table 4-8. Wind Direction Percentage Summary for meteorological stations on the coast of the Continental US.**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Station** | **North** | **South** | **East** | **West** |
| **East Coast** |
| Miami FL | 24 | 23 | 40 | **12\*** |
| Vero Beach FL | 15 | 21 | 31 | **15\*** |
| Charleston SC | 24 | 25 | 20 | **25\*** |
| Wilmington NC | 24 | 22 | 18 | **25\*** |
| Norfolk VA | 25 | 26 | 21 | **24\*** |
| Atlantic City NJ | 26 | 22 | 14 | **29\*** |
| Boston MA | 26 | 17 | 17 | **39\*** |
| Portland ME | 30 | 24 | 13 | **31\*** |
| **West Coast** |
| San Diego CA | 33 | 18 | **8\*** | 33 |
| Santa Barbara CA | **11\*** | 23 | 17 | 26 |
| San Francisco CA | 25 | 13 | **10\*** | 48 |
| Astoria OR | 15 | 26 | **25\*** | 28 |

Source: EPA’s SCRAM website (<http://www3.epa.gov/ttn/scram/surfacemetdata.htm>).

North – 304 to 34 degrees, East – 34 to 124 degrees, South – 124 to 214 degrees, West – 214 to 304 degrees

\*Winds blow from inland toward the ocean. Due to calm winds (winds < 1 m/s), where wind directions can be uncertain, percentages may not sum to 100%.

*Inhalation exposures to hatchlings and adults*

Toxicity data are not available for inhalation exposures involving reptiles nor are data available for the typical surrogate taxa (*i.e.*, birds). In several acute inhalation studies with laboratory rats, no mortality is observed at 1 mg a.i./L-air, which is equivalent to 120 mg a.i./kg-bw (MRIDs 42307236, 43665605, 42993303). The relative sensitivities of reptiles and mammals are unknown, leading to uncertainty associated with the representativeness of the available toxicity data for sea turtles. Based on the limited available data, inhalation exposure is not considered to be of concern.

**7.1.4. Effects Determinations**

The effect determination is “likely to adversely affect” (LAA) based on exposures in near shore saltwater habitats (for all listed sea turtles) and freshwater habitats (for the green sea turtle) for direct effects through dietary exposure and indirect effects through impacts to prey. There is also some concern for risk due to dermal exposures resulting from spray drift transport to adult and juvenile turtles on beaches. Although there is a high degree of overlap of exposure estimates and thresholds and other toxicity endpoints (*i.e.,* “high risk”), the confidence associated with the LAA determination is low due to uncertainties associated with the available toxicity data and the exposure estimates. In cases where critical habitat has been designated for a listed sea turtle, an LAA determination is also made for the critical habitat, based on the same considerations for direct and indirect effects (**Table 4-9**).

**Table 4-9.** **Summary of the Effects Determinations for Diazinon and Listed sea turtles and Their Designated Critical Habitat(s).**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| ***Scientific Name*** | **Common Name** | **Listing Status\*** | **FWS/NMFS Species ID (ENTITY\_ID)** | **Risk (Direct Effects)** | **Confidence (Direct Effects)** | **Risk (Indirect Effects)** | **Confidence (Indirect Effects)** | **Species Call?** | **Critical Habitat Call?** |
| *Caretta caretta* | Loggerhead sea turtle (North Pacific Ocean DPS) | E | 9941 | High | Low | High | Low | LAA | NA\*\* |
| *Caretta caretta* | Loggerhead sea turtle (Northwest Atlantic Ocean DPS) | T | 9707 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (Central North Pacific DPS) | T |  10485 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (Central South Pacific DPS) | E |  11175 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (Central West Pacific DPS) | E |  11176 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (East Pacific DPS) | T |  11191 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (North Atlantic DPS) | T |  11192 | High | Low | High | Low | LAA | LAA |
| *Chelonia mydas* | Green sea turtle (South Atlantic DPS) | T |  11193 | High | Low | High | Low | LAA | LAA |
| *Dermochelys coriacea* | Leatherback sea turtle | E | 154 | High | Low | High | Low | LAA | LAA |
| *Eretmochelys imbricata* | Hawksbill sea turtle | E | 153 | High | Low | High | Low | LAA | LAA |
| *Lepidochelys kempii* | Kemp's ridley sea turtle | E | 155 | High | Low | High | Low | LAA | NA\*\* |
| *Lepidochelys olivacea* | Olive ridley sea turtle (Mexican nesting population) | E | 5989 | High | Low | High | Low | LAA | NA\*\* |
| *Lepidochelys olivacea* | Olive ridley sea turtle | T | 160 | High | Low | High | Low | LAA | NA\*\* |

\*E = endangered, T = threatened

\*\*Not applicable because critical habitat has not been designated for this species.

## Whale and Deep Sea Fish Analysis

**7.2.1. Cetaceans**

There are currently 12 cetaceans found in the U.S. that are federally listed as endangered or threatened or that are candidates for listing (*i.e., Balaena mysticetus, Balaenoptera borealis, Balaenoptera edeni, Balaenoptera musculus, Balaenoptera physalus, Delphinapterus leucas, Eubalaena glacialis, Eubalaena japonica, Megaptera novaeangliae, Orcinus orca, Physeter microcephalus(=icrocephalus),* and *Pseudorca crassidens*) (see **Table 4-10** and **ATTACHMENT 1-13** for details). Four of these species have designated critical habitat(s) (*i.e.*, *Delphinapterus leucas, Eubalaena glacialis*, *Eubalaena japonica,* and *Orcinus orca*). All of these species are large mammals (ranging from 1,500 to 320,000 lb) and are found entirely in marine environments. Because EPA does not currently have methods available to adequately estimate potential exposures to listed cetaceans, potential risks to listed cetaceans are assessed qualitatively. Additionally, because it is not possible with current methodologies to adequately estimate exposures for these species, the effects determinations will be based on weighting for general risks and confidence (*i.e.*, it is not possible to apply weights for risk and confidence associated with each specific line of evidence).

**TABLE 4-10. Listed Cetacean Species (Found in the US).**

| **Scientific Name** | **Common Name** | **Size (Adult) (lb)** | **Diet** | **Habitat** |
| --- | --- | --- | --- | --- |
| *Balaena mysticetus* | Bowhead whale | 150,000 – 200,000 | Invertebrates (*e.g.*, krill, copepods, amphipods) and fish | Circumpolar (summer in ice-free waters adjacent to the Arctic Ocean, and are otherwise associated with sea ice) |
| *Balaenoptera borealis* | Sei whale | 100,000 | Invertebrates (*e.g.*, copepods, krill, and cephalopods) and small-schooling fish | Subtropical to subpolar waters; usually found in deeper waters of oceanic areas far from the coastline |
| *Balaenoptera edeni* | Bryde’s whale | 90,000 | Plankton (like krill and copepods), crustaceans (like red crabs and shrimp), schooling fish (like anchovies, herring, mackerel, pilchards, and sardines) | Circumglobal (tropical, subtropical, and temperate ocean waters from 40o South to 40o North) |
| *Balaenoptera musculus* | Blue whale | 320,000 | Krill and pelagic crabs | Marine waters (primarily offshore distribution) |
| *Balaenoptera physalus* | Fin whale | 165,000 | Pelagic crustaceans (*e.g.*, krill) and schooling fish (*e.g.*, herring) | Marine waters (globally) |
| *Delphinapterus leucas* | Beluga whale (Cook Inlet DPS)\* | 3,000 - 3,300 | Marine invertebrates (*e.g.*, crabs, shrimp, clams) and marine and anadromous fish (*e.g.*, cod and salmon) | Circumpolar (found in shallow coastal waters; spend the ice-free months in the upper Cook Inlet, and expand their distribution south into more offshore waters of the middle Cook Inlet) |
| *Eubalaena glacialis* | North Atlantic Right Whale\* | 140,000 | Copepods and other small invertebrates (*e.g.*, krill) | Primarily found in coastal or shelf waters in temperate to subarctic latitudes, but may go into deeper waters |
| *Eubalaena japonica* | North Pacific Right Whale\* | 140,000 | Zooplankton (*i.e.*, copepods, euphausids, and cyprids) | Shallow coastal waters. Though movements over deep waters are known to occur |
| *Megaptera novaeangliae* | Humpback whale | 77,000 | Small schooling fish (*e.g.*, herring) and large zooplankton (*e.g.*, krill) | Marine waters over and near edges of continental shelves throughout all ocean basins |
| *Orcinus orca* | Killer whale (Southern resident DPS)\* | 8,400 - 12,275 | Fish (*e.g.*, Pacific salmon and herring), squid, and marine mammals) – obligate with Pacific salmon | Coastal marine waters (Strait of Georgia, Strait of Juan de Fuca, and Puget Sound) |
| *Physeter microcephalus(=icrocephalus)* | Sperm whale | 30,000 – 90,000 | Large animals that occupy deep waters of the ocean (*e.g.*, squid, sharks, and skates) | Temperate and tropical waters with depths > ~2,000 ft (preference for deep waters) |
| *Pseudorca crassidens* | false killer whale (Main Hawaiian Islands Insular DPS) | 1,500 | Fishes (*e.g.*, tuna) and cephalopods | Waters within 140 km of the main Hawaiian islands |

\*Has designated critical habitat.

Direct effects to listed cetaceans from diazinon are not expected due to dilution in the marine environments (very low potential for exposure) and the cetaceans’ very large size (very low potential for effects). Additionally, some of the listed cetaceans are found primarily in deep, ocean waters [*i.e.*, Sei whale (*Balaenoptera borealis*), Bryde’s whale (*Balaenoptera edemi*), blue whale (*Balaenoptera musculus*), fin whale (*Balaenoptera physalus*), humpback whale (*Megaptera novaeangliae*), and sperm whale (*Physeter microcephalus(=icrocephalus*)], and/or are circumpolar [*i.e.*, the bowhead whale (*Balaena mysticetus*)]. Species that are found primarily in deep waters or are circumpolar (*i.e.,* found at high latitudes around the earth’s Polar Regions) are expected to range far from any potential application sites – further limiting the potential for exposure. In addition, since diazinon is readily metabolized and do not accumulate in aquatic organisms (see **section 6.1.1**), dietary exposure for these species is of very low concern (see **APPENDIX 3-1**). Therefore, for direct effects, the risk is considered low (due to limited exposure and potential for effects) and the confidence is considered high. The same conclusions and rationale apply to the designated critical habitats associated with these species (see **Table 4-11**).

For indirect effects (*i.e.*, reductions in whales’ prey), due to the effect of dilution in the types of marine environments in which the listed cetaceans are found and distance from potential use sites, risks from the potential loss of marine invertebrate and vertebrate prey are not expected. Therefore, for the listed cetaceans that rely wholly on marine prey [*i.e.*, bowhead whale, Bryde’s whale, Sei whale, blue whale, fin whale, North Atlantic right whale, North Pacific right whale, humpback whale, sperm whale, false killer whale (Main Hawaiian Islands Insular DPS)], indirect effects from the potential loss of prey are not expected. For these species the risk for indirect effects is considered low (due to limited exposure) and the confidence is considered high. The same conclusions and rationale apply to the designated critical habitats associated with these species. Therefore, a Not Likely to Adversely Affect (NLAA) effects determination is made for the bowhead whale, Bryde’s whale, Sei whale, blue whale, fin whale, North Atlantic right whale, North Pacific right whale, humpback whale, sperm whale, false killer whale (Main Hawaiian Islands Insular DPS), and for the designated critical habitat associated with the North Atlantic Right Whale DPS and the North Pacific Right Whale DPS.

There are currently two listed species of cetaceans that feed on both marine and anadromous prey items - Beluga whale (Cook Inlet DPS) and the killer whale (Southern resident DPS). The Beluga whale relies on a variety of aquatic invertebrate and vertebrate prey items. Many of its prey species are wholly marine, while some of its fish prey are anadromous species. While many Beluga whales are circumpolar, the Cook Inlet DPS of Beluga whales is found primarily in the Cook Inlet (off the Gulf of Alaska in Southcentral Alaska). These Beluga whales are found in both shallow coastal areas and in deeper waters, depending on the time of year. Although, there are some potential diazinon use sites found in Southcentral Alaska, they are limited and largely removed from coastal areas. Therefore, the likelihood that exposures will reach the estuarine/marine environments at concentrations high enough to impact a large marine mammal, such as a Beluga whale, is expected to be very low. Furthermore, as stated earlier, since diazinon is readily metabolized and does not accumulate in aquatic organisms, dietary exposure for this species is of very low concern. Therefore, direct effects from the use of these chemicals are not expected. Additionally, the Beluga whales rely on several prey items (most of which are wholly marine), and, the potential use sites are limited and largely removed from coastal areas. Because of these factors, for the Beluga whale the risk for indirect effects is considered low (due to limited exposure potential) and the confidence is considered high. The same conclusions and rationale apply to the designated critical habitat associated with this species. Therefore, a Not Likely to Adversely Affect (NLAA) effects determination is made for the Beluga whale and its designated critical habitat.

The killer whale (Southern resident DPS), is found in the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, and is an obligate with Pacific salmon (which are anadromous). As discussed previously, direct effects to listed killer whales are not expected. However, because the Pacific salmon, on which the killer whales depend, may be exposed to and adversely impacted by diazinon in the Pacific Northwest before reaching the marine environment there is a potential for indirect effects to this species due to a loss of prey items. Because the listed killer whales are obligates with Pacific salmon, the potential risk associated with a loss of this prey item is considered high, and the confidence is considered high. The same conclusions and rationale apply to the designated critical habitat associated with this species. Therefore, a Likely to Adversely Affect (LAA) determination is made for the killer whale (Southern resident DPS) and its designated critical habitat from the use of diazinon based on indirect effects (*i.e.*, potential impacts to prey).

**Table 4-11. Summary of the Effects Determinations for Diazinon and Listed Cetaceans and Their Designated Critical Habitat(s).**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **Scientific Name** | **Common Name** | **Listing Status\*** | **FWS/NMFS Species ID (ENTITY\_ID)** | **Risk (Direct Effects)** | **Confidence (Direct Effects)** | **Risk (Indirect Effects)** | **Confidence (Indirect Effects)** | **Species Call?** | **Critical Habitat Call?** |
| *Balaena mysticetus* | Bowhead whale | E | 3133 | Low | High | Low | High | NLAA | NA\*\* |
| *Balaenoptera borealis* | Sei whale | E | 1769 | Low | High | Low | High | NLAA | NA\*\* |
| *Balaenoptera edemi* | Bryde’s whale | C | 178 | Low | High | Low | High | NLAA | NA\*\* |
| *Balaenoptera musculus* | Blue whale | E | 3199 | Low | High | Low | High | NLAA | NA\*\* |
| *Balaenoptera physalus* | Fin whale | E | 3096 | Low | High | Low | High | NLAA | NA\*\* |
| *Delphinapterus leucas* | Beluga whale (Cook Inlet DPS) | E | 10144 | Low | High | Low | High | NLAA | NLAA |
| *Eubalaena glacialis* | North Atlantic Right Whale | E | 2510 | Low | High | Low | High | NLAA | NLAA |
| *Eubalaena japonica* | North Pacific Right Whale | E | 10145 | Low | High | Low | High | NLAA | NLAA |
| *Megaptera novaeangliae* | Humpback whale | E | 5623 | Low | High | Low | High | NLAA | NA\*\* |
| *Orcinus orca* | Killer whale (Southern resident DPS) | E | 9126 | Low | High | High | Medium | LAA | LAA |
| *Physeter microcephalus(=icrocephalus)* | Sperm whale | E | 4719 | Low | High | Low | High | NLAA | NA\*\* |
| *Pseudorca crassidens* | false killer whale (Main Hawaiian Islands Insular DPS) | E | 10700 | Low | High | Low | High | NLAA | NA\*\* |

\*E = endangered; C = candidate species

\*\*Not applicable because critical habitat has not been designated for this species.

**7.2.2. Sharks**

There is currently one listed shark species (including three DPSs found in US waters) [*i.e.*, the scalloped hammerhead shark (Eastern Pacific DPS; Central and Southwest Atlantic DPS, and Indo-West Pacific DPS) (*Sphyrna lewini*)] and three candidate shark species found in the US [*i.e.*, the cusk shark (*Brosme brosme*); oceanic whitetip shark (*Carcharhinus longimanus*); and porbeagle shark (*Lamna nasus*)] (**Table 4-12**). None of these species have designated critical habitat. All four of the species (including all three scalloped hammerhead shark DPSs) are found entirely in marine environments. Because EPA does not currently have methods available to adequately estimate potential exposures to ocean-dwelling species, potential risks to listed sharks are assessed qualitatively, and the effects determinations will be based on weighting for general risks and confidence (*i.e.*, it is not possible to apply weights for risk and confidence associated with each specific line of evidence).

**TABLE 4-12. Listed and Candidate Shark Species (Found in the US).**

|  |  |  |  |
| --- | --- | --- | --- |
| **Scientific Name** | **Common Name** | **Diet** | **Habitat** |
| *Sphyrna lewini* | Scalloped hammerhead shark (Eastern Pacific DPS) | Crustacean, teleosts, cephalopods, and rays | Coastal pelagic species that can also be found in ocean waters (to depths up to 1,000 m); found in temperate and tropical seas |
| Scalloped hammerhead shark (Central and Southwest Atlantic DPS) |
| Scalloped hammerhead shark (Indo-West Pacific DPS) |
| *Brosme brosme* | Cusk shark | Crustaceans, fish, and echinoderms | Marine; deep water with depths > 100m and water temperatures of 30 to 50oF |
| *Carcharhinus longimanus* | Oceanic whitetip shark | Fish, sea turtles, sea birds, gastropods, squid, crustaceans, and mammalian carrion (dead whales and dolphins) | Usually found offshore in waters >600 ft deep |
| *Lamna nasus* | Porbeagle shark | Fish and squids | Usually found offshore |

Direct effects to the listed and candidate sharks from diazinon are not expected due to dilution in the marine environments (very low potential for exposure). Additionally, the cusk, oceanic whitetip, and porbeagle sharks are only (or primarily) found in deep waters and, thus, are expected to range far from any potential application sites – further limiting the potential for exposure. In addition, since diazinon is readily metabolized and does not accumulate in aquatic organisms, dietary exposure for shark species is of very low concern. Therefore, for direct effects, the risk is considered low (due to limited exposure and potential for effects) and the confidence is considered high (see **Table 4-13**).

For indirect effects (*i.e.*, reductions in sharks’ prey), due to the effect of dilution in the types of marine environments in which the listed sharks are found, risks from the potential loss of marine invertebrate and vertebrate prey are not expected. Therefore, for the listed and candidate sharks, which rely wholly on marine prey, indirect effects from the potential loss of prey are not expected. For these species the risk for indirect effects is considered low (due to limited exposure) and the confidence is considered high. Therefore, a Not Likely to Adversely Affect (NLAA) effects determination is made for the scalloped hammerhead shark (Eastern Pacific DPS; Central and Southwest Atlantic DPS; and Indo-West Pacific DPS), the cusk shark, the oceanic whitetip shark, and the porbeagle shark.

**Table 4-13. Summary of the Effects Determinations for Diazinon and Listed and Candidate Sharks.**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **Scientific Name** | **Common Name** | **Listing Status\*** | **ID number** | **Risk (Direct Effects)** | **Confidence (Direct Effects)** | **Risk (Indirect Effects)** | **Confidence (Indirect Effects)** | **Species Call?** | **Critical Habitat Call?** |
| *Sphyrna lewini*  | Scalloped hammerhead shark (Eastern Pacific DPS) | E | 10733 | Low | High | Low | High | NLAA | NA\*\* |
| *Sphyrna lewini*  | Scalloped hammerhead shark (Central and Southwest Atlantic DPS) | E | 10734 | Low | High | Low | High | NLAA | NA\*\* |
| *Sphyrna lewini*  | Scalloped hammerhead shark (Indo-West Pacific DPS) | T | 10736 | Low | High | Low | High | NLAA | NA\*\* |
| *Brosme brosme* | Cusk shark | C | NMFS137 | Low | High | Low | High | NLAA | NA\*\* |
| *Carcharhinus longimanus* | Oceanic whitetip shark | C | NMFS175 | Low | High | Low | High | NLAA | NA\*\* |
| *Lamna nasus* | Porbeagle shark | C | NMFS176 | Low | High | Low | High | NLAA | NA\*\* |

\*E = endangered, T = threatened, C = candidate

\*\*Not applicable because critical habitat has not been designated for this species.

## Marine Mammals (excluding Whales) Analysis

This assessment considers the effects of diazinon on 11 listed species of seals, sea lion, and walrus (pinnipeds), sea otters (mustelids), a manatee (sirenid), and the polar bear (**Table 4-14**). The biological information (*e.g.,* diet, habitat) used in this assessment for each species is included in Attachment 1-13. This assessment does not consider listed whales (which were addressed above in **section 6.2**).

**Table 4-14. Listed pinniped, mustelid and other marine mammal species considered in this analysis.**

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| ***Scientific Name*** | **Common Name** | **Listing Status\*** | **Critical Habitat?** | **FWS/NMFS Species ID (ENTITY\_ID)** |
| *Arctocephalus townsendi* | Guadalupe fur seal | T | No | 3318 |
| *Enhydra lutris kenyoni* | Northern sea otter (Southwest Alaska DPS) | T | Yes | 5232 |
| *Enhydra lutris nereis* | Southern sea otter | T | No | 45 |
| *Erignathus barbatus* | Bearded Seal (Beringia) | T | No | 10381 |
| *Eumetopias jubatus* | Steller sea lion (Western DPS) | E | Yes | 7115 |
| *Neomonachus schauinslandi* | Hawaiian monk seal | E | Yes | 2891 |
| *Odobenus rosmarus ssp. Divergens* | Pacific walrus | C | No | 9709 |
| *Phoca largha* | Spotted seal (Southern DPS) | T | No | NMFS182 |
| *Phoca vitulina richardii* | Pacific harbor seal (Iliamna lake) | C | No | NMFS159 |
| *Trichechus manatus* | West Indian Manatee | E | Yes | 7 |
| *Ursus maritimus* | Polar bear | T | No | 8861 |

\*E=endangered; T=threatened, C = candidate

Species specific exposures to diazinon are determined based on habitat use. All of the species considered here utilize marine habitats, especially to forage. Manatees and Steller sea lions forage in freshwater, as well as marine, habitats. While in aquatic habitats, exposure to diazinon via contaminated dietary items is assessed. Other exposure routes (*e.g.,* dermal) are not assessed in aquatic habitats because methods are not available to estimate exposures via non-dietary routes. When considering the pinnipeds and mustelids, all 9 listed species consume benthic invertebrates (*e.g., cephalopods*, crabs) as well as fish. In addition, one species (Steller sea lion) eats birds and marine mammals. The pacific walrus also eats marine mammals (seals). The polar bear primarily consumes marine mammals, such as seals. The manatee is unique among the marine mammals in that it is an herbivore, consuming algae and aquatic plants. With the exception of the manatee, all of the species assessed here also utilize terrestrial habitats (*e.g.,* for breeding, as haul-outs, etc.). Exposure routes of concern in terrestrial habitats include inhalation and dermal interception of spray droplets on the day of the application. **Table 4-15** summarizes the diets and habitats of the species included in this assessment.

**Table 4-15. Summary of diets and habitats of listed marine mammals included in this assessment.**

|  |  |  |
| --- | --- | --- |
| **Listed species** | **Habitat(s)** | **Diet** |
| Seals (pinnipeds) and sea otters (mustelids) | terrestrial (haul-outs), intertidal nearshore, subtidal nearshore  | Benthic invertebrates, fish |
| Steller sea lion and Pacific walrus (pinnipeds) | terrestrial (haul-outs), intertidal nearshore, subtidal nearshore, and offshore marine | Benthic invertebrates, fish, marine mammals, birds |
| Polar bear | terrestrial, intertidal nearshore, subtidal nearshore, and offshore marine, offshore marine | Marine mammals |
| Manatee | freshwater habitats, marine nearshore, subtidal nearshore, offshore marine | Algae, aquatic plants |

This document provides the methods used to assess exposures and potential effects of diazinon to marine mammals (excluding whales) in their aquatic and terrestrial habitats. Indirect effects to prey and habitat are also assessed in aquatic habitats. Effects determinations for the species and their designated critical habitats are based on the risk assessments that integrate the weight of evidence based on exposure, available effects data and the uncertainties associated with these data.

**7.3.1. Risk Assessment for Aquatic Habitats: Direct Effects**

*Dietary toxicity data*

As indicated above, dietary exposure is assessed for aquatic habitats. Dose-based mammalian thresholds reported in Chapter 2 are converted to dietary based values[[9]](#footnote-9) using standard laboratory conversion factors (6.67x for mice and 20x for rats, WHO 2009[[10]](#footnote-10)). These values are included in **Table 4-16**. These values are used in combination with BCF values for aquatic prey in order to calculate aquatic concentrations that would constitute a potential risk from dietary exposure. In this approach, the threshold or endpoint (dietary based) is divided by the BCF. Since a BCF represents the ratio of the chemical concentration in animal tissue (mg a.i./kg-diet) to the concentration in water (mg a.i./L-water), this approach assumes that the threshold is equivalent to the chemical concentration in animal tissue.

**Table 4-16. Dietary based endpoint values calculated from dose-based thresholds and endpoints.**

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Threshold or endpoint** | **Effect** | **Dose-based Value (mg a.i./kg-bw)** | **Dietary-based value (mg a.i./kg-food)\*** | **Test species** | **Comments** | **Source\*\*** |
| Direct (1/million) | Mortality | 2.38 | 15.9 | Mouse (*Mus musculus*) | Calculated from LD50 = 105 ug/g bwAnd Slope = 2.89 | E85110 |
| Indirect (1/10) | Mortality | 37.8 | 252 |
| LD50 | Mortality | 105 | 700 |
| Direct | Sublethal (AChE inhibition) | 0.35 | 7.0 | Rat (*Rattus norvegicus*) | Based on Comparative cholinesterase assayBMDL10= 0.35 mg/kg bw/dayBMD10=0.52 mg/kg bw/day | MRID 46166302 |
| Indirect | Sublethal (AChE inhibition) | 0.52 | 10 |
| Direct and indirect | Reproduction NOEC | 0.5 | 10 | Rat | None | MRID 41158101  |
| Reproduction LOEC | 5 | 100 |

\*Calculated using standard laboratory conversion factors of 6.67x for mice and 20x for rats (WHO 2009).

\*\*See chapter 2 for additional information on these data.

When considering bioaccumulation in aquatic organisms, pesticide uptake occurs through two routes: respiration of water and consumption (dietary) of contaminated food items. Bioaccumulation is limited by the extent to which a pesticide is eliminated (through respiration or fecal elimination) or metabolized to non-toxic degradates. For chemicals with Log Kow < 4, exposure from food becomes insignificant because uptake and depuration through respiration controls the residue in the organism. For chemicals with Log Kow > 6, diet is the predominant uptake route, with little influence from respiration. Given that the Log Kow value of diazinon is 3.77, Bioconcentration Factors (BCFs), which are based on studies involving respiratory uptake only, are considered representative of bioaccumulation because the dietary route is not substantial relative to respiration. Diazinon is not expected to accumulate over time in prey because it is not persistent in aqueous environments (**Chapter 3**), and is readily excreted and metabolized by animals[[11]](#footnote-11),[[12]](#footnote-12). BCFs for diazinon in aquatic plants, invertebrates and fish are used predict short term uptake of diazinon in aquatic organisms that represent the prey of listed marine mammals. Diazinon exposures to listed marine mammals that eat air breathing animals that consume fish or other aquatic organism (*e.g.,* birds, seals) are considered discountable because they are not expected to take up diazinon through respiration, diazinon does not accumulate through the diet, and once they consume diazinon through prey, it is expected that it will be metabolized by the birds and mammals.

As noted in **Chapter 3** (exposure characterization), the KABAM[[13]](#footnote-13)-estimated BCF for plants is 280. This value is uncertain because it is based on a model estimate that does not account for metabolism of diazinon by plants. For aquatic invertebrates and fish, several empirically based values are available from different studies (7 invertebrate species, 10 fish species), leading to increased confidence in available values. Minimum, maximum and mean empirical BCF values for aquatic invertebrates and fish are provided in **Table 4-17**. These BCFs are used to estimate aqueous concentrations sufficient to exceed the thresholds and endpoints where effects are observed (*i.e.,* lowest LC50 and reproduction LOEC). Given that these BCF values are based on steady state, it is necessary to compare EECs that are representative of the time to steady state for diazinon. Based on the Log Kow of diazinon (3.7), steady state is reached in approximately 4 days. Therefore, in this assessment, 4-day average water concentrations are compared to the values in **Table 4-17**.

**Table 4-17. Aqueous concentrations defining dietary exposures of concern for marine mammals (excluding whales).**

|  |  |  |  |
| --- | --- | --- | --- |
| **Food item** | **Description of BCF** | **BCF value** | **Aqueous concentration (µg a.i./L) above which there is concern for dietary exposure** |
| **Mortality threshold** | **Lowest LC50** | **Sublethal threshold** | **Lowest NOEC for reproduction** | **Lowest LOEC for reproduction** |
| Aquatic plants | KABAM estimate | 280 | 57 | 2500 | 25 | 36 | 357 |
| Aquatic invertebrates | lowest of empirical | 3 | 5,300 | 230,000 | 2333 | 3,300 | 33,000 |
| mean of empirical | 25 | 640 | 28,000 | 280 | 400 | 4,000 |
| highest of empirical | 82 | 190 | 8,500 | 85 | 122 | 1,200 |
| Fish | lowest of empirical | 18 | 880 | 39,000 | 389 | 560 | 5,600 |
| mean of empirical | 88 | 180 | 8,000 | 80 | 110 | 1,100 |
| highest of empirical | 213 | 75 | 3,300 | 33 | 47 | 470 |

There are notable uncertainties in using the available toxicity data for mammals to represent those of pinnipeds, mustelids, manatees and bears. The available toxicity data are primarily based on rodent test species. No data are available for orders represented by the marine mammal species considered in this assessment. There is uncertainty in the relative sensitivities of laboratory rats and mice to the assessed species.

*Exposures in estuaries and near shore areas*

There is a great deal of uncertainty in estimating potential diazinon exposures in estuaries and near-shore areas of the ocean because the existing fate and transport models do not account for water stratification, tidal flux, and complex currents that occur in these habitats (*i.e.*, bins 8 and 9). As noted in the problem formulation (**Chapter 1**), estimates developed for aquatic bin 2 are generally used as surrogate exposure levels for intertidal nearshore waterbodies (bin 8), and estimates developed using aquatic bin 3 are generally used as surrogate exposure levels for subtidal nearshore waterbodies (bin 9). Additionally, aquatic bin 5 is generally used as a surrogate for tidal pools occurring during low tide (aquatic bin 8).

EECs for bins 2, 3, and 5, which include both runoff and drift, are 186-2040, 6.23-315, and 236-4250 µg a.i./L, respectively (EECs are presented in **Appendix 3-4b**; values based on HUCs 1-3, 8, 12, 13, 17-21, which are relevant to the species ranges). These 1-in-15 year 4-day average EECs exceed the mortality and sublethal thresholds (**Table 4-17**) for marine mammals that ingest aquatic plants, invertebrates and fish. Therefore, there is concern for potential direct effects to marine mammals exposed via diet (aquatic plants, invertebrates and fish) while in estuaries and near shore habitats. Most of the dietary items are not expected to contain concentrations that would exceed the lowest available LD50 or reproduction LOEC for mammals (**Table 4-17)**. Therefore, the risk is considered medium.

It should be noted that there is a great deal of uncertainty surrounding the EECs for listed pinnipeds, mustelids and manatees. The marine bins selected for these species (bins 8 and 9) cannot be modeled, so surrogate freshwater bins (2, 3, and 5) are used. Bins 2 and 3 are used to represent low and high tide periods, with bin 5 representing a tidal pool. The EECs for these bins reflect contributions from both runoff and spray drift from treated areas adjacent to the waterbodies, which may not typically occur for intertidal and subtidal nearshore waterbodies which have beaches between the treated area and the leading edge of the water. Fate data (*e.g.*, hydrolysis, metabolism) are not available for diazinon in saltwater environments, so it is uncertain how representative the EECs are for a marine environment. While the flowing bins (bins 2 and 3) are being used to represent the exchange of water expected in the marine bins due to the tides, there is uncertainty in how well these bins reflect the turbulent, mixing nature of the 12-hour tidal cycle and the EECs that may be present. The surrogate bins have lower depths and widths than the bins they are designed to represent: a 0.1 m depth and 1-2 m width for bins 2 and 5 compared to 0.5 m depth and 50 m width for bin 8 and a 1 m depth and 8 m width for bin 3 compared to 5 m depth and 50 m width for bin 9. While the smaller bins (bins 2 and 5) could be used for rearing juveniles, it is uncertain if these surrogate bins could hold sufficient volume to contain adult pinnipeds, mustelids and manatees. The additional depth and width assigned to bins 8 and 9 could also result in lower EECs based on the additional water volume and hence dilution.

Some monitoring data are available where diazinon concentrations are measured in estuaries. Concentrations reported in CA in 2008-2009 are <1µg a.i./L[[14]](#footnote-14). Measured data include documented detections; however, values are below the aqueous concentrations of concern in **Table 4-17**. It should be noted that the utility of these data are limited in that they represent ambient monitoring data for which the applications of diazinon in the watershed of the sampled estuaries are not defined and therefore the monitoring data are not expected to capture peak exposures.

EECs resulting from drift only transport into estuaries are provided in **Tables 4-4** (maximum ground spray applications of 4 lb a.i./A) **and 4-5** (maximum aerial application of 2 lb a.i./A). Exceedances of thresholds vary by bin and distance. EECs for water bodies located near treated fields are most likely to exceed sublethal and mortality thresholds.

*Exposures in Off-Shore Habitats*

It is expected that, given the large volume of water in oceans, and the lack of persistence of diazinon, this chemical will be sufficiently diluted to not be of concern for marine mammals in deep water ocean habitats. In addition, since diazinon is readily metabolized and does not accumulate in aquatic organisms, exposure via consumption of aquatic plants, invertebrates, fish and marine mammals is not of concern.

It should be noted that this approach differs from that taken above for smaller, near shore habitats (*e.g.,* estuaries). In these areas, short term concentration of diazinon in aquatic prey of marine mammals is considered because concentrations may be higher in water and diazinon could concentrate for a short term period in prey of marine mammals. The use of empirically based bioconcentration factors allows for consideration of metabolism and for uptake in prey (fish, invertebrates) via the major route of exposure (*i.e.,* respiration).

*Exposures in freshwater environments (for Manatee and Steller sea lion only)*

Aquatic bins that are used as surrogates for estimating exposures to manatees and Steller sea lions in freshwater habitats are bins 3 and 4 (Attachment 1-10). The 1-in-15 year 4-day average EECs, which include both runoff and drift, derived for these bins are on the order of 6.23-315 µg a.i./L and 4.72-351 µg a.i./L, respectively (Bin 3 and 4 EECs are presented in **Appendix 3-4b**). These 1-in-15 year 4-day average EECs exceed the concentrations that represent sublethal (25 µg a.i./L) and mortality (57 µg a.i./L) thresholds listed in **Table 4-17**. These concentrations also exceed the lowest reproduction NOEC and LOEC values (36 and 360 µg a.i./L). Therefore, there is concern (high risk) for potential direct effects to manatees exposed via diet while in freshwater habitats.

**7.3.2. Risk Assessment for Aquatic Habitats: Indirect Effects**

**Table 4-18** summarizes the thresholds used to assess indirect effects to marine mammals through impacts to diet (aquatic plants, invertebrates and fish) or habitat (plants). Details of how these values were derived are provided in Chapter 2.

**Table 4-18. Thresholds for indirect effects (µg a.i./L).**

|  |  |  |
| --- | --- | --- |
| **Taxa** | **Mortality threshold** | **Sublethal threshold** |
| Aquatic plants | Not applicable | 3700 |
| Aquatic invertebrates | 0.259 | 0.42 |
| Fish | 123.5 | 0.47 |

For estuaries and near shore ocean habitats, 1-in-15 year average daily EECs estimated for bins 2, 3, and 5 are on the order of 15-6070 µg a.i./L (EECs for bins 2, 3, and 5 are presented in Table 3-10). These values are above thresholds for mortality and sublethal thresholds for aquatic plants, aquatic invertebrates and fish (**Table 4-18**). Therefore, there is concern for indirect effects to marine mammals in near shore habitats.

For marine mammals in the off shore habitats, indirect effects due to loss of prey or habitat are not expected. This is due to the effect of dilution in deep water ocean environments in which the mammals are found.

For freshwater habitats used by the manatee and Steller sea lion, 1-in-15 year average daily EECs estimated for bins 3 and 4 are on the order of 14-947 µg a.i./L. These values are below the threshold for aquatic plants, therefore, effects to plant dietary items and habitat are not expected. These values are above the thresholds for fish and aquatic invertebrates, which represent prey of the Steller sea lion. Therefore, there is potential for indirect effects to the Steller sea lion through impacts to available prey in freshwater habitats.

**7.3.3. Risk Assessment for Terrestrial Habitats**

This section considers potential exposures to marine mammals (excluding whales) in terrestrial habitats. Relative to risks due to dietary exposures in aquatic habitats, risks in terrestrial habitats are expected to be lower.

*Dermal exposures*

Dermal exposure due to spray drift transport is of potential concern for mammals in terrestrial areas adjacent to treated sites. Four studies examined toxicity related to dermal exposure in mammals, including the mouse (*Mus musculus*) (Sogorb *et al.* 1993, E90688), rat (*Rattus norvegicus*) (Nichol *et al*. 1983, E88385), rabbit (*Oryctolagus cuniculus*) (Yehia *et al.* 2007, E100112), and cow (*Bos taurus*) (Danielson and Golsteyn 1997, E84471). Effects included 100% mortality in mice (Sogorb *et al.* 1993, E90688); differences in cell count, biochemistry, and relative organ weight in the rabbit (Yehia *et al.* 2007, E100112); and differences in cholinesterase activity in the cow (Danielson and Golsteyn 1997, E84471). No effects are seen on feeding efficiency or weight gain in the cow (Danielson and Golsteyn 1997, E84471) or on biochemistry in the rat (Nichol *et al*. 1983, E88385). Study durations ranged from five days to 100 days. The dermal exposure levels are reported in the following units: ppm, %, mg/10L and µg. Since these units could not be translated to environmentally relevant units (*e.g.,* mg a.i./kg-bw); the relationship between these toxicity data and exposure levels is unknown. Spray drift deposition cannot be quantitatively assessed with the available toxicity data.

There is uncertainty associated with concluding that dermal exposures of diazinon will pose a risk to marine mammals. For instance, the likelihood of exposure on a given day of application, is dependent upon the wind direction, *i.e.*, the pesticide must be transported by wind blowing from the application site toward the beach. **Table 4-8** provides a summary of the percentage of time the wind is blowing from a certain direction. These data suggest that prevailing winds blow from inland toward the ocean up to 30% of the time (independent of when diazinon is likely to be applied).

*Inhalation exposures*

In several acute inhalation studies with laboratory rats, no mortality is observed at 1 mg a.i./L-air (equivalent to 106 µg a.i./mg3) (MRIDs 42307236, 43665605, 42993303). This concentration is assumed to be orders of magnitude above what is expected in the air of marine mammals in terrestrial habitats (monitoring data from a study conducted near orchards in CA (Rider 2010) reported a maximum concentration of 4.3 µg a.i./mg3, other data cited in **Chapter 3** are orders of magnitude lower). As a result, inhalation exposure is not considered to be of concern. As noted above, there is uncertainty associated with the available surrogate test species and their relative sensitivities to the assessed species. In addition, there is uncertainty in that sublethal effects (lethargy) were observed in the registrant-submitted studies; however, NOECs were not established.

**7.3.4. Effects Determinations**

The effect determinations for the Guadalupe fur seal, southern sea otter, Steller sea lion, Hawaiian monk seal, and Pacific harbor seal are “likely to adversely affect” (LAA) based on exposures in near shore saltwater habitats for direct effects through dietary exposure and indirect effects through impacts to prey. The determination for the West Indian Manatee is also LAA based on potential direct effects in saltwater and freshwater habitats used by this species. Although there is overlap of exposure estimates and thresholds, there is little overlap with endpoints where effects are observed (*i.e.*, the risk conclusion is “medium”). In addition, the confidence associated with the LAA determination is low due to uncertainties associated with the available toxicity data and the exposure estimates. In cases where critical habitat has been designated (*i.e.*, for Steller sea lion, Hawaiian monk seal and West Indian Manatee), an LAA determination is also made for the critical habitat, based on the same considerations for direct and indirect effects (**Table 4-19**).

Several of the species included in this assessment only occur in waters of the US and terrestrial areas that are in Alaska. These species include the northern sea otter, the bearded seal, the pacific walrus, the spotted seal, and the polar bear. When considering USDA’s Census of Agriculture data for Alaska (2012), a very limited amount of land was used for orchards (17 A), vegetables (1059 A), berries (42 A) and nurseries (55 A). Most of these crops are grown in the interior of the state (*e.g.*, near Fairbanks). Although, there are some potential diazinon use sites found in Southcentral Alaska, they are limited and largely removed from coastal areas[[15]](#footnote-15). Therefore, the likelihood that exposures will reach the estuarine/marine environments at concentrations high enough to impact a large marine mammal, is expected to be very low. Therefore, exposure to diazinon is considered unlikely and NLAA determinations are made for these five species (**Table 4-19**). For the designated critical habitat of the northern sea otter, an NLAA determination is also made.

**Table 4-19.** **Summary of the Effects Determinations for Diazinon and Listed Marine Mammals (excluding whales) and Their Designated Critical Habitats.**

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| ***Scientific Name*** | **Common Name** | **Listing Status\*** | **FWS/NMFS Species ID (ENTITY\_ID)** | **Risk (Direct Effects)** | **Confidence (Direct Effects)** | **Risk (Indirect Effects)** | **Confidence (Indirect Effects)** | **Species Call?** | **Critical Habitat Call?** |
| *Arctocephalus townsendi* | Guadalupe fur seal | T | 3318 | Medium | Low | High | Low | LAA | NA\*\* |
| *Enhydra lutris kenyoni* | Northern sea otter (Southwest Alaska DPS) | T | 5232 | Low | High | Low | High | NLAA | NLAA |
| *Enhydra lutris nereis* | Southern sea otter | T | 45 | Medium | Low | High | Low | LAA | NA\*\* |
| *Erignathus barbatus* | Bearded Seal (Beringia) | T | 10381 | Low | High | Low | High | NLAA | NA\*\* |
| *Eumetopias jubatus* | Steller sea lion (Western DPS) | E | 7115 | Medium | Low | High | Low | LAA | LAA |
| *Neomonachus schauinslandi* | Hawaiian monk seal | E | 2891 | Medium | Low | High | Low | LAA | LAA |
| *Odobenus rosmarus ssp. Divergens* | Pacific walrus | C | 9709 | Low | High | Low | High | NLAA | NA\*\* |
| *Phoca largha* | Spotted seal (Southern DPS) | T | NMFS182 | Low | High | Low | High | NLAA | NA\*\* |
| *Phoca vitulina richardii* | Pacific harbor seal (Iliamna lake) | C | NMFS159 | Medium | Low | High | Low | LAA | NA\*\* |
| *Trichechus manatus* | West Indian Manatee | E | 7 | Medium | Low | Low | High | LAA | LAA |
| *Ursus maritimus* | Polar bear | T | 8861 | Low | High | Low | High | NLAA | NA\*\* |

\*E = endangered, T = threatened, C = candidate

\*\*Not applicable because critical habitat has not been designated for this species.

## Cave Dwelling Invertebrate Species Analysis

Currently, there are 22 terrestrial invertebrate species that are only found in caves; all of them only use terrestrial environments and rely wholly on terrestrial food sources [*i.e.*, Kauai cave wolf or pe'e pe'e maka 'ole spider, Coffin Cave mold beetle, Helotes mold beetle, Robber Baron Cave meshweaver, Madla's Cave meshweaver, Braken Bat Cave meshweaver, Government Canyon Bat Cave meshweaver, Tooth Cave spider, Government Canyon Bat Cave spider, Clifton Cave beetle, Icebox Cave beetle, Tatum Cave beetle, Louisville Cave beetle, Ground beetle [unnamed (Rhadine exilis)], Ground beetle [unnamed (Rhadine infernalis)], Tooth Cave ground beetle, Kauai cave amphipod, Tooth Cave pseudoscorpion, Kretschmarr Cave mold beetle, Cokendolpher Cave harvestman, Bee Creek Cave harvestman, and Bone Cave harvestman*].*  None of these species are known to have obligate relationships. Eleven of the species have designated critical habitats [*i.e.*,Kauai cave wolf or pe'e pe'e maka 'ole spider, Helotes mold beetle, Robber Baron Cave meshweaver, Madla's Cave meshweaver, Braken Bat Cave meshweaver, Government Canyon Bat Cave meshweaver, Tooth Cave spider, Ground beetle [unnamed (*Rhadine exilis*)], Ground beetle [unnamed (*Rhadine infernalis*)], Kauai cave amphipod, and Cokendolpher Cave harvestman].

Because EPA does not currently have methods available to precisely estimate potential exposures to listed cave-dwelling terrestrial invertebrates, potential risks are assessed qualitatively and characterized using exposure estimates for potential forage items outside the cave. The likelihood of exposure is dependent in part upon foraging behavior (*e.g.*, subterranean only versus inside and outside the cave) and the potential for transport of contaminated forage or substrates into the cave. Additionally, although none of the listed species of cave-dwelling terrestrial invertebrates discussed in this section have aquatic life stages, it is uncertain whether and to what extent they may be exposed to pesticide residues in water which percolate through permeable soils in karst cave systems. Thus, the following effects determinations are based on weighting for general risks and confidence (*i.e.*, it is not possible to apply weights for risk and confidence associated with each specific line of evidence).

For animals found only in cave interiors, exposure via spray drift is not likely due to multiple drift interception areas typically associated with caves (both outside and within cave systems). Additionally, potential runoff events that could transport pesticides to the interior of a cave system are not expected to result in significant exposure to terrestrial animals found within most caves. Therefore, direct pesticide exposures to most cave-dwelling terrestrial invertebrates from pesticide applications made outside of their cave systems are not generally expected. The exception is for those species that are found in cave systems with permeable substrates (*e.g.*, karst cave formations or sink holes) where pesticide contamination via runoff and recharge is more likely if pesticides are used in the vicinity of the cave systems. All of the currently listed cave-dwelling terrestrial invertebrates live in such environments, *i.e.*, karst systems, sinkholes, or similar subterranean habitats overlain by permeable substrates) (see **ATTACHMENT 1-20**). For all of these species, pesticides have been listed as a cause of concern by the US Fish and Wildlife Service (*e.g.*, see <http://ecos.fws.gov/docs/federal_register/fr3497.pdf>).

For terrestrial cave-dwelling species, there is also a potential for exposure to pesticides from the ingestion of food items that originate from outside of a cave. Based on the available information (see the species profiles, **ATTACHMENT 1-20** **Supplemental Information 3**for details), all of the currently listed terrestrial cave-dwelling invertebrate species appear to rely, to some degree, on dietary items that originate from exterior sources (see **Table 4-20**).

**Table 4-20. Listed Cave-Dwelling Terrestrial Invertebrates and Their Diets.**

| **Scientific Name** | **Common Name** | **Diet** | **Origin of the Dietary Item(s)** |
| --- | --- | --- | --- |
| *Adelocosa anops* | Spider, Kauai cave wolf or pe'e pe'e maka 'ole | Kauai cave amphipod, other cave-inhabiting arthropods, and alien species of arthropods that enter the cave system. | Exterior to the cave; Within the cave |
| *Batrisodes texanus* | Beetle, Coffin Cave mold | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Batrisodes venyivi* | Beetle, Helotes mold | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Cicurina baronia* | Meshweaver, Robber Baron Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Cicurina madla* | Meshweaver, Madla's Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Cicurina venii* | Meshweaver, Braken Bat Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Cicurina vespera* | Meshweaver, Government Canyon Bat Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Neoleptoeta myopica* | Spider, Tooth Cave | Examples of nutrient sources include leaf litter fallen or washed in, animal droppings, and animal carcasses | Exterior to the cave |
| *Neoleptoneta microps* | Spider, Government Canyon Bat Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Pseudoanophthalmus caecus* | Beetle, Clifton Cave | Small invertebrates and cave cricket eggs | Exterior to the cave; Within the cave |
| *Pseudoanophthalmus frigidus* | Beetle, Icebox Cave | Small invertebrates and cave cricket eggs | Exterior to the cave; Within the cave |
| *Pseudoanophthalmus parvus* | Beetle, Tatum Cave | Small invertebrates and cave cricket eggs | Exterior to the cave; Within the cave |
| *Pseudoanophthalmus troglodytes* | Beetle, Louisville Cave | Small invertebrates and cave cricket eggs | Exterior to the cave; Within the cave |
| *Rhadine exilis* | Ground beetle, [unnamed] | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Rhadine infernalis* | Ground beetle, [unnamed] | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Rhadine persephone* | Beetle, Tooth Cave ground | A variety of troglobites are known to feed on cave cricket eggs, feces, and/or on the adults and nymphs directly. | Exterior to the cave; Within the cave |
| *Spelaeorchestia koloana* | Kauai cave amphipod | Roots of *Pithecellobium dulce* (Manila tamarind) and *Ficus* sp. (fig), rotting roots, sticks, branches, and other plant material washed into, or otherwise carried into caves, as well as the fecal material of other arthropods | Exterior to the cave; Within the cave  |
| *Tartarocreagris texana* | Pseudoscorpion, Tooth Cave | Examples of diet nutrient sources for these species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Texamaurops reddelli* | Beetle, Kretschmarr Cave mold | Examples of nutrient sources include leaf litter fallen or washed in, animal droppings, and animal carcasses | Exterior to the cave |
| *Texella cokendolpheri* | Harvestman, Cokendolpher Cave | Examples of diet nutrient sources for this species include leaf litter fallen or washed in, animal droppings, and animal carcasses. | Exterior to the cave |
| *Texella reddelli* | Harvestman, Bee Creek Cave | Leaf litter, animals’ droppings and carcasses. | Exterior to the cave |
| *Texella reyesi* | Harvestman, Bone Cave | Leaf litter, animals’ droppings and carcasses. | Exterior to the cave |

Kauai cave wolf spidersmay ingest some arthropods that originate from outside of the cave, however, its diet is primarily made up of the Kauai cave amphipod (and other cave-inhabiting arthropods). For the Kauai cave amphipod, roots of terrestrial plants that extend into subterranean habitats serve as a primary food source, however, some nutritional sources that originate from outside the cave (*e.g.*, washed in vegetative matter and feces) may also be ingested. Additionally, the Tooth Cave ground beetle is assumed to eat primarily cave crickets (eggs, nymphs, and adults). Cave cricket adults may forage outside of their cave system at night. In addition to cave cricket eggs, Clifton Cave, Icebox Cave, Tatum Cave, and Louisville Cave beetles eat cave invertebrates which originate primarily, but perhaps not exclusively, from within their caves. Therefore, Kauai cave wolf spiders, Kauai cave amphipods, and Tooth Cave ground beetles, Clifton Cave beetles, Icebox Cave beetles, Tatum Cave beetles, and Louisville Cave beetles rely primarily, but not exclusively, on food items that originate from within their cave systems. Thus, there is a potential for exposure from the ingestion of contaminated prey/forage items, but current methods which assume that 100% of the diet is contaminated are not adequate for estimating such exposures.

Although the potential for exposure (*i.e.*, the chance of a contaminated food item is washed into a cave and then ingested) is likely low, because diazinon is a broad spectrum insecticide that is highly toxic to a wide range of invertebrates. The potential for effects to a single individual of Kauai cave wolf spiders, Kauai cave amphipods, Tooth Cave ground beetles, Clifton Cave beetles, Tatum Cave beetles, and Louisville Cave beetlescannot be precluded if they ingest a dietary item(s) that is contaminated with one of these pesticides (see analysis below). Therefore, for direct effects to Kauai cave wolf spiders, Kauai cave amphipods, Tooth Cave ground beetles, Clifton Cave beetles, Tatum Cave beetles, and Louisville Cave beetles from the use of diazinon, the potential for risk is ‘medium’ (*i.e.*, there is a potential for effects if exposure occurs, but it’s not clear what effects would occur at the exposure concentrations), and the confidence is ‘low’ (*e.g.*, potential exposures cannot be estimated, but they also cannot be precluded). The same is true for indirect effects related to the loss of potential prey items (that originate from outside of the caves). The same conclusions and rationale apply to the designated critical habitats associated with these species. Therefore, for Kauai cave wolf spiders, Kauai cave amphipods, Tooth Cave ground beetles, Clifton Cave beetles, Tatum Cave beetles, and Louisville Cave beetles a Likely to Adversely Affect (LAA) determination is made for diazinon. Additionally, for the designated critical habitats associated with Kauai cave wolf spiders, Kauai cave amphipods, and Tooth Cave ground beetles a Likely to Adversely Affect (LAA) determination is made for diazinon. An NLAA call is made for the potential direct and indirect effects to Icebox Cave beetles because this species only overlaps with diazinon use on cattle ear tags and effects from that use are considered discountable.

The remaining listed cave-dwelling terrestrial invertebrate species appear to rely primarily on food items that are derived from exterior sources (*i.e.*, leaf litter, animal droppings, and carcasses that may fall or be washed into their cave systems). Therefore, there is a potential for exposure if their food item(s) is contaminated with a pesticide. The EPA does not currently have methods for estimating the concentration of pesticides in leaf litter, animal feces, or carcasses that are found within cave systems. Again, because diazinon is a broad spectrum insecticide that is highly toxic to a wide range of invertebrates, the potential for effects to a single individual of a cave-dwelling invertebrate cannot be precluded if it ingests a contaminated food item.

For leaf litter, the foliar dissipation half-lives for diazinon is estimated to be 5.3 days (Chapter 3). However, because of the sensitivity of terrestrial invertebrates to diazinon, the EECs on leaves are estimated to be high enough to exceed effects thresholds for weeks after spraying. For diazinon, there are no terrestrial invertebrate mortality data based on dietary exposure (mg a.i./kg-diet), however there are dietary data available for sublethal thresholds (*i.e.*, behavior). The sublethal LOAEC, based on behavior, is exceeded for 20 days and 34 days based on diazinon application rates of 0.5 and 3 lb a.i./acre, respectively. For the animal food items, the daily fraction retained in mammals and birds is 0.1 and 0.214, respectively, for diazinon. Therefore, if a food item is contaminated with diazinon outside of the cave, it could contain residues high enough to cause concern for several days, if not weeks, after exposure. Thus, there is a potential for exposure to and effects from diazinon to cave-dwelling invertebrates from the ingestion of contaminated leaf litter and carcasses.

Additionally, there is evidence in the literature indicating that animal feces (*e.g.*, guano) and carcasses contaminated with pesticides (including some organophosphates) have been found in cave systems (*e.g.*, Eidels, *et al.* 2007[[16]](#footnote-16); Land 2001[[17]](#footnote-17); MacFarland 1998[[18]](#footnote-18); and Sandel 1999[[19]](#footnote-19)). Therefore, the potential for exposure is medium and the potential for exposure from the ingestion of contaminated feces and carcasses cannot be precluded. Furthermore, diazinon is highly toxic to a wide range of invertebrates. Therefore, for direct effects to Coffin Cave mold beetle, Helotes mold beetle, Robber Baron Cave meshweaver, Madla's Cave meshweaver, Braken Bat Cave meshweaver, Government Canyon Bat Cave meshweaver, Tooth Cave spider, Government Canyon Bat Cave spider, Ground beetle [unnamed (Rhadine exilis)], Ground beetle [unnamed (Rhadine infernalis)], Tooth Cave pseudoscorpion, Kretschmarr Cave mold beetle, Cokendolpher Cave harvestman, Bee Creek Cave harvestman, and Bone Cave harvestman*,* from the use of diazinon, the potential for risk is ‘medium’ (*i.e.*, there is a potential for effects if exposure occurs, but it’s not clear what effects would occur at the exposure concentrations), and the confidence is ‘medium’ (*e.g.*, potential exposures cannot be estimated, but they also cannot be precluded). The same conclusions and rationale apply to the designated critical habitat associated with these species.

For potential indirect effects, diazinon use would have the potential to impact the availability of animal feces (by impacting the animals), carcasses, and leaf litter (by impacting terrestrial plants. The chance that the impacts on the availability of animal feces, carcasses, and leaf litter would reach a level that would adversely affect the listed terrestrial invertebrates due to the loss of food, is very low, but cannot be precluded. Therefore, for indirect effects to Coffin Cave mold beetle, Helotes mold beetle, Robber Baron Cave meshweaver, Madla's Cave meshweaver, Braken Bat Cave meshweaver, Government Canyon Bat Cave meshweaver, Tooth Cave spider, Government Canyon Bat Cave spider, Ground beetle [unnamed (Rhadine exilis)], Ground beetle [unnamed (Rhadine infernalis)], Tooth Cave pseudoscorpion, Kretschmarr Cave mold beetle, Cokendolpher Cave harvestman, Bee Creek Cave harvestman, and Bone Cave harvestman from the use of diazinon, the potential for risk is ‘medium’ (*i.e.*, there is a potential for effects if exposure occurs, but it’s not clear what effects would occur at the exposure concentrations), and the confidence is ‘low’ (*e.g.*, potential exposures cannot be estimated, but they also cannot be precluded). The same conclusions and rationale apply to the designated critical habitat associated with this species.

Therefore, based on the potential for direct and indirect effects, for Coffin Cave mold beetle, Helotes mold beetle, Robber Baron Cave meshweaver, Madla's Cave meshweaver, Braken Bat Cave meshweaver, Government Canyon Bat Cave meshweaver, Tooth Cave spider, Government Canyon Bat Cave spider, Ground beetle [unnamed (Rhadine exilis)], Ground beetle [unnamed (Rhadine infernalis)], Tooth Cave pseudoscorpion, Kretschmarr Cave mold beetle, Cokendolpher Cave harvestman, Bee Creek Cave harvestman, and Bone Cave harvestman, a Likely to Adversely Affect (LAA) determination is made for diazinon.

Additionally, for the designated critical habitat associated withMadla's Cave meshweaver, Ground beetle [unnamed (Rhadine exilis)], and Ground beetle [unnamed (Rhadine infernalis)], a Likely to Adversely Affect (LAA) determination is made for diazinon. NLAA calls are made for the designated critical habitats for two species, Helotes mold beetle and Braken bat cave meshweaver because their critical habitats only overlap with diazinon use on cattle ear tags and effects from that use are considered discountable. NE calls are made for the designated critical habitats for four species, including Robber baron cave meshweaver, Government canyon bat cave meshweaver, Government canyon bat cave spider and Cokendolpher cave harvestman.

**Table 4-21. Summary of the Effects Determinations for Diazinon and Listed Terrestrial Cave-Dwelling Invertebrates and Their Designated Critical Habitat(s).**

| **Species name** | **Common name** | **Listing Status\*** | **FWS Species ID (ENTITY\_ID)** | **Risk (Direct Effects)** | **Confidence (Direct Effects)** | **Risk (Indirect Effects)** | **Confidence (Indirect Effects)** | **Species Call?** | **Critical Habitat Call?** |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| *Adelocosa anops* | Spider, Kauai cave wolf or pe'e pe'e maka 'ole | E | 463 | Medium | Low | Medium | Low | **LAA** | **LAA** |
| *Batrisodes texanus* | Beetle, Coffin Cave mold | E | 447 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Batrisodes venyivi* | Beetle, Helotes mold | E | 460 | Medium | Medium | Medium | Low | **LAA** | **NLAA****\*\*\*** |
| *Cicurina baronia* | Meshweaver, Robber Baron Cave | E | 472 | Medium | Medium | Medium | Low | **LAA** | **NE** |
| *Cicurina madla* | Meshweaver, Madla's Cave | E | 471 | Medium | Medium | Medium | Low | **LAA** | **LAA** |
| *Cicurina venii* | Meshweaver, Braken Bat Cave | E | 474 | Medium | Medium | Medium | Low | **LAA** | **NLAA****\*\*\*** |
| *Cicurina vespera* | Meshweaver, Government Canyon Bat Cave | E | 473 | Medium | Medium | Medium | Low | **LAA** | **NE** |
| *Neoleptoneta myopica* | Spider, Tooth Cave | E | 467 | Medium | Medium | Medium | Low | **LAA** | **LAA** |
| *Neoleptoneta microps* | Spider, Government Canyon Bat Cave | E | 470 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Pseudoanophthalmus caecus* | Beetle, Clifton Cave | C | 5064 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Pseudoanophthalmus frigidus* | Beetle, Icebox Cave | C | 2862 | Low | High | Low | High | **NLAA\*\*\*** | NA\*\* |
| *Pseudoanophthalmus parvus* | Beetle, Tatum Cave | C | 7134 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Pseudoanophthalmus troglodytes* | Beetle, Louisville Cave | C | 3379 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Rhadine exilis* | Ground beetle, [unnamed] | E | 461 | Medium | Medium | Medium | Low | **LAA** | **LAA** |
| *Rhadine infernalis* | Ground beetle, [unnamed] | E | 459 | Medium | Medium | Medium | Low | **LAA** | **LAA** |
| *Rhadine persephone* | Beetle, Tooth Cave ground | E | 449 | Medium | Low | Medium | Low | **LAA** | NA\*\* |
| *Spelaeorchestia koloana* | Kauai cave amphipod | E | 485 | Medium | Low | Medium | Low | **LAA** | **LAA** |
| *Tartarocreagris texana* | Pseudoscorpion, Tooth Cave | E | 466 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Texamaurops reddelli* | Beetle, Kretschmarr Cave mold | E | 448 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Texella cokendolpheri* | Harvestman, Cokendolpher Cave | E | 469 | Medium | Medium | Medium | Low | **LAA** | **NE** |
| *Texella reddelli* | Harvestman, Bee Creek Cave | E | 464 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |
| *Texella reyesi* | Harvestman, Bone Cave | E | 465 | Medium | Medium | Medium | Low | **LAA** | NA\*\* |

\*E = endangered

\*\*NA = Not applicable (*i.e.*, there is no critical habitat designated for this species).

\*\*\*See cattle ear tag analysis (Appendix 4-4)

## Cattle Ear Tag Use Analysis

Diazinon is used in cattle ear tags to control insect pests. Species of interest to this assessment were identified by overlapping ranges and designated critical habitats of listed species with sites where cattle ear tags could be used. Due to the unique application of the pesticide in a slow release plastic device applied to a cow’s ear, the same methodology used to assess other use patterns is not applicable to this use pattern and therefore the cattle ear tag analysis was completed as a separately (**APPENDIX 4-4**).

For diazinon, there are areas where no other uses overlap with the spatial footprint represented by cattle ear tags. Effects determinations are made for those species with ranges and critical habitats (if designated) that are within the diazinon action area but the only overlap is with the spatial extent of the cattle ear tag use.

# Refined Risk Analysis for 11 Listed Bird Species: TIM-MCnest Analysis

One of the major recommendations of the National Academy of Sciences was to utilize probabilistic methods for assessing risks of pesticides to listed species. A refined analysis was conducted to explore the utility of currently available probabilistic, refined methods for use in biological evaluations of listed species. There is also potential utility for use of these methods in the biological opinions for these species. Only a subset of species were selected to explore applications of the models that may inform future method development and to identify data needs. A detailed description of this refined analysis is included in **APPENDIX 4-7**.

Two refined risk assessment models available for birds were used in this analysis, including the Terrestrial Investigation Model (TIM) and the Markov Chain Nest Productivity Model (MCnest) TIM estimates the probability and magnitude of mortality to exposed birds. MCnest estimates declines in fecundity associated with exposure. Both models incorporate species-specific life history parameters (*e.g.,* diet), pesticide use information (*e.g.,* crop, application rate), fate data (*e.g.,* foliar dissipation half-life) and toxicity data. These models integrate toxicity data for mortality, growth, reproduction and behavioral effects.

**Appendix 4-7** also includes a proof of concept analysis that involves estimating the number of individuals within a population that are potentially exposed to diazinon. One species, the least bell’s vireo was selected for this analysis. The method relies upon known life history characteristics of the listed species, in particular, habitat usage and considers potential overlap of these habitats with diazinon use sites and habitats potentially receiving spray drift.

*TIM results*

TIM was run to examine the likelihood of mortality to birds exposed to diazinon from spray drift from orchard crops, ground fruit and vegetables and nurseries. When median estimates of the most sensitive parameters are used (*i.e.,* LD50 = 1.51 mg a.i./kg-bw and foliar dissipation half-life = 2.2 d), the following conclusions can be drawn:

* For orchards, there is high probability (83.4% or greater) of mortality to an exposed individual of all species simulated, with the exception of the masked bobwhite, which has a medium probability (28.1%) of mortality to an individual.
* For ground fruit and vegetables, there is a high probability (89.1% or greater, >99.9% for most species) of mortality to an exposed individual for all simulated species.
* For nurseries, there is high probability (80.1% or greater) of mortality to an exposed individual of all species simulated, with the exception of the masked bobwhite Atwater’s prairie chicken, which have a medium probability (24.4 and 61.1%, respectively) of mortality to an individual.

This provides additional evidence to the effects determinations in that if an individual is exposed, there is a high likelihood of mortality.

Probability distribution functions for different uses were compared to investigate the relative risks associated with different uses. The least bell’s vireo was used as an example to illustrate the utility of this analysis. For this species, almost all birds exposed to diazinon from applications to lettuce likely to die (approximately 99%). Risks to birds exposed to diazinon applied to other ground fruit (*e.g.,* tomatoes), orchards and nurseries is substantially lower, with an estimated magnitude of 60-80 for tomatoes, 40-60% mortality for almond orchards and 15-35% for nurseries. This information is considered useful in estimating the magnitude of effect in the portion of the population that is exposed and may be used in evaluating potential population-level impacts of diazinon on listed birds.

*MCnest results*

When considering impacts to fecundity, mortality is the major contributor to declines; however, reproductive effects do occur in surviving birds. In general, applications made after the breeding period occur do not result in substantial declines in fecundity because exposed adults are able to successfully reproduce prior to exposures that lead to mortality. Two examples are provided here that illustrate impacts to fecundity of birds: the yellow billed cuckoo and least bell’s vireo.

*Estimated number of individual least bells vireos potentially exposed to diazinon*

Based on 5 years of land cover data for potential diazinon use sites and 9 years of national-level diazinon usage data, the average estimated number of individuals exposed per year is 107 individual least bell’s vireos (4% of the population). When considering an upper bound of the number of individuals exposed based on maximum estimates of usage and acreage of potential diazinon use sites, as many as 221 least Bell’s vireos (8% of the population) may be exposed in a given year. Aerial applications to lettuce are most likely contributing to the majority of the exposed individuals. When considering the TIM results, it is most likely that the majority (99%) of these exposed individuals will die. When considering impacts to fecundity, if applications occur during the breeding season (*i.e.,* spring and summer), mortality in adult birds leads to substantial declines in number of broods and fledglings per female. If adult birds survive diazinon exposures, there is potential for reproductive effects. There is uncertainty associated with the number of estimated Least Bell’s vireos exposed to diazinon in a given (due to application of national level usage data, some of which may not be reflective of current diazinon labels). When more recent, state-level usage data from CA PUR is used, the estimated number of individuals exposed per year is an order of magnitude lower (*i.e.,* 11).

This analysis was conducted to explore methods for estimating the number of individuals exposed using currently available data and to identify data needs. Major uncertainties associated with this approach include: assumptions of uniform distributions of individuals within a population, use of national level usage data from 9 years to represent potential usage over the next 15 years and lack of consideration of exposure during migration. GIS data providing information of distributions (and densities) of individuals during migration and the breeding season would be helpful to address the first two uncertainties.

# Additional Information relevant for Step 3: ECx and Web-ICE Analyses

Two additional analyses were conducted that may provide relevant information for the Step 3 process, though they did not factor into the weight of evidence approach employed for Step 2. The first analysis was an evaluation of available chronic toxicity data for diazinon for use in estimating ECx values based on regression models in the Toxicity Response Analysis Program (TRAP) v1.22. More details on this analysis can be found in **ATTACHMENT 4-2**.

The second analysis was an evaluation of the Web-based Interspecies Correlation Estimation (Web-ICE) tool for use in assessing risks to listed species. This evaluation consisted of two separate analyses. The first evaluated the ability of Web-Ice to predict toxicity values for each listed species of fish using data for standard test specie (*i.e.,* rainbow trout, bluegill sunfish, and fathead minnow. While Web-ICE is able to predict direct toxicity of a chemical to 17 species of listed fish, the majority of listed species would rely on the availability of genus and family level models for toxicity predictions. An analysis of genus and family level models indicates that when these models were developed from the most sensitive value for each chemical they were generally protective of the most sensitive species within predicted taxa, including listed species, and were more protective than geometric means models. The second analysis explored the appropriateness of using tools such as Web-ICE to develop SSDs, in cases where empirical datasets for particular taxa are limited, by comparing empirically derived SSDs for chemicals with large datasets to SSDs derived from predicted datasets. More details on these analyses can be found in **ATTACHMENT 4-3**.

1. Federal Geographic Data Committee. FGDC-STD-001-1998. Content standard for digital geospatial metadata (revised June 1998). Federal Geographic Data Committee. Washington, D.C. [↑](#footnote-ref-1)
2. Hawaii. Dept. of Business, Economic Development and Tourism. Research and Economic Analysis Division. Statistics and Data Support Branch. State of Hawaii data book; a statistical abstract. Honolulu: 2006. [↑](#footnote-ref-2)
3. There are differences between the species in habitat use by life stage and some are primarily offshore during certain stages. For example, leatherback and olive ridley turtles spend the majority of their time as juveniles and adults in the open ocean and use the nearshore habitats far less frequently than other species. Habitat descriptions for each species can be found at http://www.nmfs.noaa.gov/pr/species/esa/listed.htm#turtles. [↑](#footnote-ref-3)
4. In a chicken metabolism study (MRID 41225901), 87% of residues were excreted after 24 hours. 85% of diazinon was metabolized to non-toxic degradates. [↑](#footnote-ref-4)
5. In a rat metabolism study (MRID 41108901), 90% of residues were excreted after 24 hours. 85% of diazinon was metabolized to non-toxic degradates. Trace amounts of diazinon were detected in excreta. [↑](#footnote-ref-5)
6. Kow (based) Aquatic BioAccumulation Model. See Chapter 3 for discussion of how the plant BCF was estimated. [↑](#footnote-ref-6)
7. The LD50 for the bullfrog is a non-definitive value of >2000 mg a.i./kg-bw reported. The mallard and pheasant LD50 values are 3.54 and 4.33 mg a.i./kg-bw, respectively. [↑](#footnote-ref-7)
8. Smalling, K.L., and Orlando, J.L. (2011). Occurrence of pesticides in surface water and sediments from three central California coastal watersheds, 2008–09: U.S. Geological Survey Data Series 600, 70 p. [↑](#footnote-ref-8)
9. Given the uncertainty in scaling the toxicity value across 2-6 orders of magnitude from tested species (0.02-0.35 kg) to assessed species (20-1620 kg), dose-based exposures are not generated. Concentration-based exposures are compared directly to available concentration-based dietary toxicity endpoints. [↑](#footnote-ref-9)
10. World Health Organization. 2009. Principles and methods for the risk assessment of chemicals in food, Annex 2, dose conversion table. Environmental health criteria 240. [↑](#footnote-ref-10)
11. In a chicken metabolism study (MRID 41225901), 87% of residues were excreted after 24 hours. 85% of diazinon was metabolized to non-toxic degradates. [↑](#footnote-ref-11)
12. In a rat metabolism study (MRID 41108901), 90% of residues were excreted after 24 hours. 85% of diazinon was metabolized to non-toxic degradates. Trace amounts of diazinon were detected in excreta. [↑](#footnote-ref-12)
13. Kow (based) Aquatic BioAccumulation Model. See Chapter 3 for discussion of how the plant BCF was estimated. [↑](#footnote-ref-13)
14. Smalling, K.L., and Orlando, J.L. (2011). Occurrence of pesticides in surface water and sediments from three central California coastal watersheds, 2008–09: U.S. Geological Survey Data Series 600, 70 p. [↑](#footnote-ref-14)
15. An overlap analysis was not conducted for aquatic species, therefore, the extent of overlap in ranges and potential diazinon use sites is not available. [↑](#footnote-ref-15)
16. Eidels, R.R., J.O. Whitaker Jr., and D.W. Sparks 2007. Insecticide residues in bats and guano from Indiana. Proceedings of the Indiana Academy of Science, 116 (1): 50 – 57. [↑](#footnote-ref-16)
17. Land, T.A. 2001. Population Size and Contaminant Exposure of Bats Using Caves on Fort Hood Military Base, Texas. M.S. thesis, Texas A&M University, College Station, Texas. [↑](#footnote-ref-17)
18. McFarland, C.A. 1998. Potential Agricultural Insecticide Exposure of Indiana Bats (*Myotis sodalis*) in Missouri. M.S. thesis, University of Missouri, Columbia, Missouri. [↑](#footnote-ref-18)
19. Sandel, J.K. 1999. Insecticides and Bridge-Roosting Colonies of Mexican Free-Tailed Bats (*Tadarida brasiliensis*) in Texas. M.S. thesis, Texas A&M University, College Station, Texas. [↑](#footnote-ref-19)