**APPENDIX 4-8. Additional Qualitative Species Analyses**

# Step 2a: Is the species exposure pathway considered incomplete?

For imidacloprid, three types of species characteristics led to a conclusion that the exposure pathway is incomplete: species that only occur on uninhabited islands, species that predominantly occur in the open ocean and terrestrial species that only occur in caves. Additional explanation of why the exposure pathway is incomplete for these three types of species habitats is provided below.

Species whose ranges only occur on uninhabited islands are not expected to be exposed to imidacloprid because imidacloprid is not reasonably expected to be applied in areas not inhabited by humans. According to imidacloprid’s usage data (**APPENDIX 1-4**) the majority of imidacloprid’s usage is on residential and agricultural uses, which would not be expected to occur on uninhabited islands. The lack of quantitative seed treatment usage data does not represent a major uncertainty for these species since this use is relevant to agricultural uses. Other registered uses of imidacloprid (**APPENDIX 1-2**) would not be expected to be needed in areas where people do not live.

Exposures to species that predominantly occur in the open ocean (*e.g.,* whales) or rely on ocean species (*e.g.,* seabirds) are reasonably expected to be *de* *minimus*. This is because imidacloprid is not applied directly to the ocean, sources of imidacloprid (runoff and spray drift) that reach the open ocean are diluted, and it does not bioaccumulate.

Imidacloprid is not registered for applications within caves. Exposures to terrestrial organisms living within caves are expected to be *de minimus*. As discussed in Chapter 3, the major transport routes of imidacloprid from treatment sites to non-target areas include spray drift and runoff. Since caves are enclosed, spray drift transport is not reasonably expected to result in exposures to cave dwelling organisms. Runoff transport may lead to imidacloprid reaching ground water that is associated with caves. Therefore, for aquatic species that inhabit caves (e.g., barton springs salamander), exposures and associated risks are assessed in Steps 2e through 2i.

For listed terrestrial species that are obligate to caves (e.g., spiders), exposure from water is expected to be *de minimus*. The atmosphere of the inner cave (where these obligate cave species live) is saturated with water vapor. Species have adapted to this hydrating environment by increasing their permeability such that they “become freshwater animals living in an aerial environment” (Howarth 1987). This means that species get the majority of their water needs met by the atmosphere and from consumption of their prey. For terrestrial obligate cave species, water sources are limited to the condensation in the cave and on cave walls resulting from groundwater sources or from detritus/guano. Imidacloprid is classified as non-volatile from dry non-adsorbing surfaces and water. As a result, imidacloprid is not expected to be presented in water vapor or condensation water that may occur in caves.

Another possible route of exposure is from leaf litter, animal droppings, and carcasses that may fall or be washed into cave systems. While there is evidence in the literature indicating that animal feces (*e.g*., guano) and carcasses contaminated with pesticides have been found in cave systems (*e.g*., Land, *et al.* 2019; Eidels, *et al*. 2012; Eidels, *et al.* 2007; Land 2001; MacFarland 1998; and Sandel 1999), imidacloprid residues in these studies were not analyzed as they focused on other pesticides (*e.g.*, organochlorines, organophosphates, carbamates). Based on the physical properties of imidacloprid, residues may not be expected because it is rapidly metabolized and excreted from the body. Therefore, exposures to species that rely on food items that are derived from exterior sources are expected to be *de minimus*.

# Step 2d: Are exposure models considered unreliable for assessed species?

At this time, the current exposure models used in this assessment do not estimate exposures for all types of pesticide applications, all habitat types, or for all potential exposure routes relevant to listed species. Therefore, there may be uncertainty in the exposure values being used for a particular species based on what potential uses its range or critical habitat may overlap with, what type of habitat the species is found in, or what the main potential exposure route(s) might be. For species and critical habitats that have not been determined to be NE or NLAA based on the above analyses, consideration is given to how well the conceptual model of the relevant exposure model(s) matches up with the specific species being assessed. If the model estimates are not considered representative of the exposure of the species (due to an inconsistency in the exposure model and assessed species’ habitat), a qualitative analysis is conducted.

The qualitative analysis considered whether exposures to imidacloprid are reasonably certain to occur given the habitat of the listed species (e.g., ocean, beach, and/or freshwater habitats) and, if exposures are expected to occur, are impacts to an individual likely. The analysis also considered the potential for effects to the prey, pollination, habitat and/or dispersal (PPHD) of the species and whether those effects would rise to the level of impacting an individual of a listed species.

### Aquatic Species

This discussion focuses primarily upon species that utilize marine and estuarine habitats. Effects to marine mammals (e.g., pinnipeds, mustelids, polar bear, manatee), sea birds, and sea turtles are considered for both aquatic and terrestrial exposures. Effects to fish and corals are considered for aquatic exposures only. Since imidacloprid is not considered bioaccumulative and is not expected to accumulate in the tissue of prey, exposure from eating contaminated fish would be very low. In the marine environment, exposure of these species to conventional pesticides is not reasonably expected to reach the estuarine/marine environments at concentrations high enough to impact an individual of a species because of dilution. Additionally, tidal reversal in freshwater streams and vertical stratification of the freshwater inflow due to differences in salinity and temperature can enhance the mixing process at the freshwater/marine interface and disperse potential pesticide concentrations that may occur in freshwater streams and rivers that discharge into marine environments, limiting the potential for a pesticide to reach individuals of the listed species. See **APPENDIX 4-1** for the complete list of species considered for exposure in the marine environment.

Marine mammals, sea birds, and sea turtles may also spend a portion of their life-cycle (*i.e.,* breeding and basking) on shore, so the potential for exposure in the terrestrial environment is also considered. See **APPENDIX 4-1** for the complete list of aquatic species with the potential for terrestrial exposure. Potential exposure routes include inhalation and dermal interception of spray droplets on the day of application. Since these species do not forage while on land, dietary exposure while in terrestrial habitats is not expected. Based on the points below, exposure at concentrations high enough to impact an individual are not reasonably expected to occur for these species.

* In a quantitative assessment, the overlap analysis assumes that all individuals of the species are in the terrestrial portion of their range, which represents a relatively small fraction of the entire range of the species. This artificially inflates the overlap numbers resulting in low confidence in the potential for exposure.
* While in the terrestrial environment, exposure of these species would be limited to spray drift from use sites adjacent to nesting or basking sites. The potential for exposure in the terrestrial environment is limited because on the day of application, imidacloprid would have to be transported by wind blowing from the application site toward the beach with little opportunity for interception of spray droplets.
* The duration of potential exposures would be limited as these species spend a relatively short amount of time on the shore for basking and/or breeding purposes. For example, sea turtles utilize beaches to lay their eggs, while some species use beaches to bask, however, sea turtles spend the vast majority of their lives in aquatic habitats.
* In addition, several of the species only occur in aquatic and terrestrial areas that are in Alaska. These species include the bearded seal, the Pacific walrus, the ringed seal, and the polar bear. Although, there are some potential pesticide use sites found in Southcentral Alaska, they are likely limited and/or largely removed from coastal areas. A limited amount of land is used for growing grains and fruits and vegetables, based on USDA’s Census of Agriculture data for Alaska (2012). Most of these crops are grown in the interior of the state (e.g*.*, near Fairbanks). Although, there are some potential agricultural use sites found in Southcentral Alaska (e.g., forage crops), they are limited and largely removed from coastal areas. Therefore, pesticide exposure to these species is not reasonably expected to occur.

Effects to the PPHD of marine mammals, fish, sea birds, sea turtles, and corals are also considered. The listed species considered rely on more than one dietary item, most of which are entirely marine. In estuarine/marine environments, exposures to conventional pesticides are not reasonably expected to decrease prey populations. Therefore, a NLAA determination is made for these species (**APPENDIX 4-1**).

Two species were given additional consideration for this exposure pathway and are discussed below. These species are the Western manatee and the killer whale.

The Western manatee forages in freshwater, as well as marine environments and requires freshwater on a regular basis. There is a great deal of uncertainty in estimating potential imidacloprid exposures in marine environments that support the Western manatee, but it is possible to use Estimated Environmental Concentrations (EECs) for the large flowing bins (3 and 4) to estimate exposures in freshwater environments (max daily EECs for HUC 3 = 1770 µg/L for Rice, all other UDLs max EECs = 26.6 µg/L). In addition, there are uncertainties in the potential for effects due to uncertainties in the toxicity database, which utilizes small mammals as a surrogate. The effects thresholds for imidacloprid are include an LD50 = 424 mg/kg-bw/day and a LOAEL = 20 mg/kg-bw/day based on decreases in food consumption and body weight of laboratory rats, which weigh several orders of magnitude less than manatees. Translating these toxicity endpoints to direct exposure in water for the manatee is difficult but using drinking water consumption as an estimate for the exposure to imidacloprid, exposure would be orders of magnitude below these toxicity thresholds[[1]](#footnote-1). Based on the Kow of imidacloprid and BCF studies (see Chapter 3), imidacloprid is also not expected to accumulate greatly in aquatic plants, which represent the diet of this species. As imidacloprid does not cause effects to aquatic plants at environmentally relevant concentrations, (IC50 = 17,600 µg/L for non-vascular aquatic plants and IC50 = 280,000 µg/L for vascular aquatic plants) effects to the PPHD of the manatee are not anticipated. Therefore, a NLAA determination is made for the Western manatee.

The killer whale (*Orcinus orca*, Southern resident DPS), is found in the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, and has an obligate relationship with Pacific salmon (which are anadromous), including several species (Chinook, Chum, and Coho) that are themselves considered threatened or endangered. Imidacloprid exposures are reasonably expected to be *de minimus* due to dilution and the fate characteristics of imidacloprid (i.e., not expected to bioaccumulate); therefore, exposures to killer whales are not expected. The obligate relationship of salmon with the killer whale is unique as species of salmon are also listed and are assessed in this BE, which allows for a more detailed analysis of the obligate species. Looking across a subset of salmon species for which effects determinations were made, the estimated percent of the population impacted ranged from ~30 to 50% impacted under the deterministic maximum scenario assessed and based entirely on potential effects to the PPHD of salmon (aquatic invertebrates), as no direct effects to fish are anticipated. In reviewing these species further, it was determined that most of these impacts were based on the poultry litter layer. As documented in Chapter 4, the application of this layer is based on conservative assumptions regarding the use and spatial extent of this layer, due to the lack of specific usage data. When salmon species are reanalyzed without the poultry litter layer, leaving use sites where the overlap is more reliable, predictions of potential impacts are greatly reduced, with the potential population impacted estimated to range from approximately <5 to 8%. Given the uncertainty around the poultry litter layer and the large reduction in the salmon population potentially impacted without this layer, as well as the basis for effects to salmon only being due to indirect effects to one part of its prey base, a LAA determination with weakest evidence is made for the killer whale (Southern resident DPS).

Of the 52 species with NLAA determinations due to incomplete exposure pathways (step 2A), 18 have designated critical habitats, also all determined to be NLAA. Additionally, of the 57 species with determinations based on unreliable exposure models (step 2D), 56 are NLAA and 1 is LAA (killer whale). Sixteen have designated critical habitats; 15 of these are NLAA determinations and 1 is a LAA determination (killer whale) (**APPENDIX 4-1**). In addition, the beluga whale occurs in waters of the US and terrestrial areas that are in Alaska. Although, there are some potential pesticide use sites found in Southcentral Alaska, they are likely limited and/or largely removed from coastal areas. A limited amount of land is used for growing grains and fruits and vegetables, based on USDA’s Census of Agriculture data for Alaska (2012). Most of these crops are grown in the interior of the state (e.g., near Fairbanks). Although, there are some potential agricultural use sites found in Southcentral Alaska (e.g., forage crops), they are limited and largely removed from coastal areas. Therefore, pesticide exposure to the critical habitat of these species is not reasonably expected to occur.

### Terrestrial Species

There is one species of terrestrial animal, the wood bison, that has extensive portions of its range located outside of the United States (i.e., in Canada). In a quantitative assessment, the overlap analysis assumes that all individuals of the species are in the portion of their range located in the United States, which represents a relatively small fraction of the entire range of the species. Since this artificially inflates the overlap numbers, which would result in low confidence in the potential for exposure, the overlap analysis was not run for these species and they are assessed qualitatively. For the wood bison, the population in the United States consists of a nonessential experimental population (NEP) established in 2015 in Western Alaska. This population is highly managed and tracked extensively. In addition, while there are some potential pesticide use sites found in Southcentral Alaska, they are likely limited and/or largely removed from areas utilized by the wood bison. A limited amount of land is used for growing grains and fruits and vegetables (USDA’s Census of Agriculture data for Alaska (2012)). Most of these crops are grown in the interior of the state (e.g., near Fairbanks). Although, there are some potential agricultural use sites found in Southcentral Alaska (e.g., forage crops), they are limited. Therefore, pesticide exposure to the wood bison is not reasonably expected to occur and a NLAA determination is made.

# References

Eidels, R.R., and J.O. Whitaker Jr. 2007. Insecticide Residues in Bats and Guano from Indiana. Proceedings of the Indiana Academy of Science 116(1):50-57.

Eidels, R.R., and J.O. Whitaker Jr. 2013. Screening of Insecticides in Bats from Indiana. Proceedings of the Indiana Academy of Science 121(2):133-142.

Howarth, F.G. 1987. The evolution of non-relictual tropical troglobites. International Journal of Speleology 16: 1-16.

Land, T.A., D.R. Clark Jr., C.E. Pekins, and T.E. Lacher Jr. 2019. Seasonal Emergence and Historical Contaminant Exposure of Cave Myotis (*Myotis velifer*) in Central Texas and Current Status of the Population. Environments; 6: 21.

McFarland, C.A. 1998. Potential Agricultural Insecticide Exposure of Indiana Bats (*Myotis sodalis*) in Missouri. Unpublished Master’s thesis.

1. Estimated freshwater drinking water consumption for manatees 145 ml/kg-bw/day (Physiological Ecology and Bioenergetics Lab, University of Central Florida. https://sciences.ucf.edu/biology/PEBL/current-research/manatee-studies/do-manatees-need-to-drink-fresh-water/). Assuming 450 kg as weight of manatee, this corresponds to 65 L/day. Using 1770 ug/L as an upper bound EEC, this equals 115 mg/day consumption of a.i. per day, which equals 0.25 mg/kg/day, much lower than toxicity thresholds. Although thresholds are typically adjusted for body weight, given the orders of magnitude difference in the test species weight (rodent) and the manatee weight, this adjustment was not applied. [↑](#footnote-ref-1)