Risks of Pentachloronitrobenzene Use to the Federally Threatened

Bay Checkerspot Butterfly (*Euphydryas editha bayensis*), California Tiger Salamander (*Ambystoma californiense*) Central California Distinct Population Segment, and Delta Smelt (*Hypomesus transpacificus*)

And the Federally Endangered

California Clapper Rail (*Rallus longirostris obsoletus*), California Freshwater Shrimp (*Syncaris pacificus*), California Tiger Salamander (*Ambystoma californiense*) Sonoma County Distinct Population Segment and Santa Barbara County Distinct Population Segment, San Francisco Garter Snake (*Thamnophis sirtalis tetrataenia*), and Tidewater Goby (*Eucyclogobius newberryi*)

Pesticide Effects Determinations
PC Code: 056502
CAS Number: 82-68-8

Environmental Fate and Effects Division
Office of Pesticide Programs
Washington, D.C. 20460

June 28, 2012

Primary Authors:
Scott Glaberman, Ph.D, Biologist
Cheryl Sutton, Ph.D., Environmental Scientist
Environmental Risk Branch IV

Secondary Review:
Thomas Steeger, Ph.D., Senior Science Advisor
Jim Carleton, Ph.D., Senior Fate Scientist
Meredith Laws, Acting Branch Chief
Environmental Risk Branch IV

Table of Contents

Appendices

Appendix A.	Verification Memo for PCNB
Appendix B.	Use Closure Memo for PCNB
Appendix C.	Risk Quotient (RQ) Method and Levels of Concern (LOCs)
Appendix D.	Example Output from PE5
Appendix E.	Example Output from T-REX and T-HERPS
Appendix F.	Summary of Ecotoxicity Data for PCNB and Degradates
Appendix G.	Bibliography of ECOTOX Open Literature
Appendix H.	Accepted ECOTOX Data Table
Appendix I.	The HED Chapter of the Reregistration Eligibility Decision Document for Pentachloronitrobenzene (Case Number 0128; Chemical Number 056502)
Appendix J.	Complete List of MRID Studies for PCNB
Appendix K.	Methods for Calculating PCNB Residues in Soil and Worms

Attachments

Attachment I:	Supplemental Information on Standard Procedures for Threatened and Endangered Species Risk Assessments on the San Francisco Bay Species
Attachment II:	Status and Life History for the San Francisco Bay Species
Attachment III:	Baseline Status and Cumulative Effects for the San Francisco Bay Species

List of Tables

List of Figures

List of Commonly Used Abbreviations and Nomenclature

µg/kg	Symbol for "micrograms per kilogram"
µg/L	Symbol for "micrograms per liter"
ºC	Symbol for "degrees Celsius"
AAPCO	Association of American Pesticide Control Officials
a.i.	Active Ingredient
AIMS	Avian Monitoring Information System
Acc#	Accession Number
amu	Atomic Mass Unit
BCB	Bay Checkerspot Butterfly
BCF	Bioconcentration Factor
BEAD	Biological and Economic Analysis Division
bw	Body Weight
CAM	Chemical Application Method
CARB	California Air Resources Board
CBD	Center for Biological Diversity
CCR	California Clapper Rail
CDPR	California Department of Pesticide Regulation
CDPR-PUR	California Department of Pesticide Regulation Pesticide Use Reporting Database
CFWS	California Freshwater Shrimp
CI	Confidence Interval
CL	Confidence Limit
CTS	California Tiger Salamander
CTS-CC	California Tiger Salamander Central California Distinct Population Segment
CTS-SB	California Tiger Salamander Santa Barbara County Distinct Population Segment
CTS-SC	California Tiger Salamander Sonoma County Distinct Population Segment
DS	Delta Smelt
EC	Emulsifiable Concentrate
EC_{05}	5% Effect Concentration
EC_{25}	25% Effect Concentration
EC_{50}	50% (or Median) Effect Concentration
ECOTOX	EPA managed database of Ecotoxicology data

EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
e.g.	Latin *exempli gratia* (“for example”)
EIM	Environmental Information Management System
EPI	Estimation Programs Interface
ESU	Evolutionarily significant unit
et al.,	Latin *et alii* (“and others”)
etc.	Latin *et cetera* (“and the rest” or “and so forth”)
EXAMS	Exposure Analysis Modeling System
FIFRA	Federal Insecticide Fungicide and Rodenticide Act
FQPA	Food Quality Protection Act
ft	Feet
GENEEC	Generic Estimated Exposure Concentration model
HPLC	High Pressure Liquid Chromatography
IC_{05}	5% Inhibition Concentration
IC_{50}	50% (or median) Inhibition Concentration
i.e.	Latin for *id est* (“that is”)
IECV1.1	Individual Effect Chance Model Version 1.1
KABAM	$\underline{K}_{OW}$ (based) <u>A</u>quatic <u>B</u>io<u>A</u>ccumulation <u>M</u>odel
kg	Kilogram(s)
kJ/mole	Kilojoules per mole
km	Kilometer(s)
K_{AW}	Air-water Partition Coefficient
K_d	Soil-water Distribution Coefficient
K_f	Freundlich Soil-Water Distribution Coefficient
K_{OC}	Organic-carbon Partition Coefficient
K_{OW}	Octanol–water Partition Coefficient
LAA	Likely to Adversely Affect
lb a.i./A	Pound(s) of active ingredient per acre
LC_{50}	50% (or Median) Lethal Concentration
LD_{50}	50% (or Median) Lethal Dose
LOAEC	Lowest Observable Adverse Effect Concentration
LOAEL	Lowest Observable Adverse Effect Level
LOC	Level of Concern
LOD	Level of Detection
LOAEC	Lowest Observable Effect Concentration

LOQ	Level of Quantitation
m	Meter(s)
MA	May Affect
MATC	Maximum Acceptable Toxicant Concentration
m^2/day	Square Meters per Days
ME	Microencapsulated
mg	Milligram(s)
mg/kg	Milligrams per kilogram (equivalent to ppm)
mg/L	Milligrams per liter (equivalent to ppm)
mi	Mile(s)
mmHg	Millimeter of mercury
MRID	Master Record Identification Number
MW	Molecular Weight
n/a	Not applicable
NASS	National Agricultural Statistics Service
NAWQA	National Water Quality Assessment
NCOD	National Contaminant Occurrence Database
NE	No Effect
NLAA	Not Likely to Adversely Affect
NLCD	National Land Cover Dataset
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observable Adverse Effect Concentration
NOAEL	No Observable Adverse Effect Level
NOAEC	No Observable Effect Concentration
NRCS	Natural Resources Conservation Service
OPP	Office of Pesticide Programs
OPPTS	Office of Prevention, Pesticides and Toxic Substances
ORD	Office of Research and Development
PCE	Primary Constituent Element
pH	Symbol for the negative logarithm of the hydrogen ion activity in an aqueous solution, dimensionless
pKa	Symbol for the negative logarithm of the acid dissociation constant, dimensionless
ppb	Parts per Billion (equivalent to µg/L or µg/kg)
ppm	Parts per Million (equivalent to mg/L or mg/kg)

PRD	Pesticide Re-Evaluation Division
PRZM	Pesticide Root Zone Model
ROW	Right of Way
RQ	Risk Quotient
SFGS	San Francisco Garter Snake
SLN	Special Local Need
TG	Tidewater Goby
T-HERPS	Terrestrial Herpetofaunal Exposure Residue Program Simulation
T-REX	Terrestrial Residue Exposure Model
UCL	Upper Confidence Limit
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WP	Wettable Powder
wt	Weight

1. Executive Summary

1.1. Purpose of Assessment

The purpose of this assessment is to evaluate potential direct and indirect effects of pentachloronitrobenzene (PCNB; PC code: 056502) on the bay checkerspot butterfly (*Euphydryas editha bayensis*) (BCB), California tiger salamander (*Ambystoma californiense*) Central California Distinct Population Segment (CTS-CC), Sonoma County Distinct Population Segment (CTS-SC) and Santa Barbara County Distinct Population Segment (CTS-SB), delta smelt (*Hypomesus transpacificus*) (DS), California clapper rail (*Rallus longirostris obsoletus*) (CCR), California freshwater shrimp (*Syncaris pacificus*) (CFWS), San Francisco garter snake (*Thamnophis sirtalis tetrataenia*) (SFGS), and the tidewater goby (*Eucyclogobius newberryi*) (TG) arising from FIFRA regulatory actions regarding use of PCNB on agricultural and non-agricultural sites. In addition, this assessment evaluates whether these actions can be expected to result in modification of designated critical habitat for the BCB, CTS-CC, CTS-SB, DS, and TG; the other assessed species do not have designated critical habitats. This assessment is in accordance with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998), procedures outlined in the Agency's Overview Document (USEPA, 2004), and consistent with a suit in which PCNB was alleged to be of concern to the BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG (*Center for Biological Diversity (CBD) vs. EPA et al.,* (Case No. 07-2794-JCS).

Below is a brief description of when each species was listed as well as a short description of the Primary Constituent Element (PCEs) (when applicable) for each San Francisco Bay species being assessed.

The BCB was listed as threatened in 1987 by the USFWS. The species primarily inhabits native grasslands on serpentine outcrops around the San Francisco Bay Area in California. The PCEs for BCBs are areas on serpentinite-derived soils that support the primary larval host plant (*i.e*., dwarf plantain) and at least one of the species' secondary host plants. Additional BCB PCEs include the presence of adult nectar sources, aquatic features that provide moisture during the spring drought and areas that provide adequate shelter during the summer diapause.

There are currently three CTS Distinct Population Segments (DPSs): the Sonoma County (SC) DPS, the Santa Barbara (SB) DPS, and the Central California (CC) DPS. Each DPS is considered separately in the risk assessment as they occupy different geographic areas. The main difference in the assessment will be in the spatial analysis. The CTS-SB was listed by the USFWS as endangered in 2000, the CTS-SC in 2002, and the CTS-CC as threatened in 2004. The CTS-SB and CTS-SC were downlisted from endangered to threatened in 2004 by the USFWS, however, the downlisting was vacated by the U.S. District Court. Therefore, the Sonoma and Santa Barbara DPSs are currently listed as endangered while the CTS-CC is listed as threatened. All CTS populations utilize vernal pools, semi-permanent ponds, and permanent ponds, and the terrestrial environment in California. The aquatic environment is essential for breeding and reproduction and mammal burrows are also important habitat for aestivation. The PCEs for the CTS are standing bodies of freshwater sufficient for the species to complete the aquatic portion of its life cycle that are adjacent to barrier-free uplands that contain small mammal burrows. An additional PCE is upland areas between sites (as described above) that allow for dispersal of the species.

The DS was listed as threatened on March 5, 1993 (58 FR 12854) by the USFWS (USFWS, 2007). DS are mainly found in the Suisun Bay and the Sacramento-San Joaquin estuary near San Francisco Bay. During spawning DS move into freshwater. The PCEs for DSs are shallow fresh or brackish backwater sloughs for egg hatching and larval viability, suitable water with adequate river flow for larval and juvenile transport, suitable rearing habitat, and unrestricted access to suitable spawning habitat.

The CCR was listed by the USFWS as an endangered species in 1970. The species is found only in California in coastal wetlands along the San Francisco estuary and Suisun Bay.

The CFWS was listed as endangered in 1988 by the USFWS. The CFWS inhabits freshwater streams in Central California in the lower Russian River drainage and westward to the Pacific Ocean and coastal streams draining into Tomales Bay and southward into the San Pablo Bay.

The SFGS was listed as endangered in 1967 by the USFWS. The species is endemic to the San Francisco Peninsula and San Mateo County in California in densely vegetated areas near marshes and standing open water.

The TG was listed as endangered in 1994 by the USFWS. The range of the TG is limited to coastal brackish water habitats along the coast of California. The PCEs for TGs are persistent, shallow aquatic habitats with salinity from 0.5 parts per thousand (ppt) to 12 ppt, that contain substrates suitable for the construction of burrows and submerged aquatic plants that provide protection. An additional PCE is the presence of sandbars that at least partially closes a lagoon or estuary during the late spring, summer, and fall.

1.2. Scope of Assessment

1.2.1. Uses Assessed

PCNB is a broad-spectrum, contact fungicide belonging to the organochlorine (polychlorinated aromatic) class of chemicals. It is used to control various diseases caused by *Rhizoctonia solani*, as well as other fungal pathogens (*e.g.*, *Botrytis* spp., *Rhizopus* spp., *Aspergillus* spp., *Pellicularia* spp., *Fusarium* spp.). The compound was registered for use in controlling fungal diseases on a variety of food/feed and non-food crops, either as a seed treatment or soil/foliar treatment, and underwent a re-registration eligibility assessment in 2005 (EPA, 2005). Although current registrations of PCNB allow for use nationwide, this ecological risk assessment and effects determination only addresses currently registered uses of PCNB in California.

PCNB is currently registered for numerous agricultural and non-agricultural uses; however, most of those uses are currently restricted. Thus, this assessment will focus only on the uses currently allowed on approved labels. This subset of uses will be known as “currently marketed uses” throughout the assessment.

PCNB is currently marketed only for use on cole crops (broccoli, Brussels sprouts, cabbage, cauliflower), golf course turf (greens, tees, fairways [GTF] only), cotton, potatoes, and ornamental bulbs (gladiolus, hyacinth, iris, narcissus, tulips, and lilies; dip/soak only), collards, kale and mustard greens. Formulation types currently approved and marketed include a flowable and a granular. Only the following currently marketed uses are allowed in California and are considered as part of the Federal action evaluated in this assessment: cole crops, golf course GTF, cotton, potatoes and ornamental bulbs (dip/soak only). For ornamental bulb dip use, an assessment was precluded by a lack of available information on use rates.

1.2.2. Environmental Fate Properties of PCNB

PCNB has low water solubility, is slightly to hardly mobile in soil, and is moderately volatile. When considered along with its polychlorinated degredates, PCNB and its degredates are expected to be persistent in the environment and to bioaccumulate in aquatic ecosystems. PCNB is stable to hydrolysis and photodegradation on soil, but photodegrades rapidly in water under optimal laboratory conditions. In the environment, however, ideal conditions for photodegradation of PCNB are expected to be limited (see Section 2.4). PCNB residues of concern (parent plus degradates) are persistent in both aerobic and anaerobic soil. Aquatic metabolism data have not been submitted for PCNB. Volatilization of PCNB and its degradates is expected to represent a relevant dissipation route, with long-range transport a possibility; however, the overall persistence of PCNB and its degradates in the air is unknown.

1.2.3. Evaluation of Degradates and Stressors of Concern

PCNB has several environmental degradates and contaminants (present in technical PCNB) that are of toxicological concern. Degradates of concern for PCNB include pentachloroaniline (PCA), pentachlorothioanisole (PCTA), pentachlorobenzene (PCB; also present as a contaminant), pentachlorophenol (PCP), pentachlorothioanisole sulfoxide (PCTASO), pentachlorothioanisole sulfone ($PCTASO_2$) and hexachlorobenzene (HCB; a contaminant). Also of concern are polychlorinated dibenzodioxin (PCDD) and polychlorinated dibenzofuran (PCDF) impurities, although these were not measured in environmental fate studies and are not accounted for in the calculated half-lives. Degradates and contaminants are of toxicological concern when their structure includes a polychlorinated phenyl ring or when they are otherwise of known toxicological concern. Section 2.2.1 provides additional information on how the PCNB residues of concern were handled in this assessment.

1.3. Assessment Procedures

A description of routine procedures for evaluating risk to the San Francisco Bay Species is provided in Attachment I.

1.3.1. Exposure Assessment

1.3.1.a. Aquatic Exposures

Tier-II aquatic exposure models were used to estimate high-end exposures of PCNB and its residues of concern in aquatic habitats resulting from runoff and spray drift from different uses. The models used to predict aquatic EECs are the Pesticide Root Zone Model coupled with the Exposure Analysis Model System (PRZM/EXAMS). Peak model-estimated environmental concentrations resulting from different PCNB uses range from 3.0 to 31.0 µg/L. A search of the California Department of Pesticide Regulation surface water database yielded no surface water monitoring data for PCNB or its major degradates PCA and PCTA.

1.3.1.b. Terrestrial Exposures

To estimate exposures to terrestrial species resulting from PCNB applications, the T-REX model is used for foliar and granular uses. The T-HERPS model is used to further characterize dietary exposures of reptiles and terrestrial-phase amphibians relative to birds. KABAM (K_{OW} (based) Aquatic Bioaccumulation Model) version 1.0 is used to estimate potential bioaccumulation of PCNB residues in an aquatic food web and subsequent risks these residues pose to organisms consuming aquatic species.

1.3.2. Toxicity Assessment

The assessment endpoints include direct toxic effects on survival, reproduction, and growth of individuals, as well as indirect effects, such as reduction of the food source and/or modification of habitat. Federally-designated critical habitat has been established for the BCB, CTS-CC, CTS-SB, DS, and TG. Primary constituent elements (PCEs) were used to evaluate whether PCNB has the potential to modify designated critical habitat. The Agency evaluated registrant-submitted studies and data from the open literature to characterize toxicity of PCNB and its degradates. The most sensitive toxicity value available from acceptable or supplemental studies for each taxon relevant for estimating potential risks to the assessed species and/or their designated critical habitat was used.

PCNB is characterized as highly toxic to freshwater fish and invertebrates, moderately toxic to estuarine/marine fish, and very highly toxic to estuarine/marine invertebrates on an acute exposure basis. Chronic exposure to PCNB resulted in reduced growth (length and weight) and reproduction (number of young produced per adult per day) in fish and aquatic invertebrates, respectively.

PCNB is practically non-toxic to adult honey bees on an acute contact exposure basis, practically non-toxic to mammals and birds on an acute oral exposure basis, and practically non-toxic to birds on a subacute dietary exposure basis. Chronic toxicity data revealed that reproduction and growth were the most sensitive endpoints in birds. Chronic toxicity testing on rats resulted in decreased growth in both parents and offspring.

No relevant aquatic or terrestrial plant toxicity data are available for PCNB or many of its degradates.

1.3.3. Measures of Risk

Risk quotients (RQs) are calculated by dividing the lowest acute and chronic taxon-specific effects concentration by the appropriate expected environmental concentration (EEC) for each use scenario. Acute and chronic RQs are compared to the Agency's Levels of Concern (LOCs) to identify instances where PCNB use has the potential to adversely affect the assessed species or adversely modify their designated critical habitat. When RQs for a particular type of effect are below LOCs, the pesticide is considered to have "no effect" on the species and its designated critical habitat. Where RQs exceed LOCs, a potential to cause adverse effects or habitat modification is identified, leading to a conclusion of "may affect". If PCNB use "may affect" the assessed species, and/or may cause effects to designated critical habitat, the best available additional information is considered to refine the potential for exposure and effects, and distinguish actions that are Not Likely to Adversely Affect (NLAA) from those that are Likely to Adversely Affect (LAA).

1.4. Summary of Conclusions

Based on the best available information, the Agency makes a Likely to Adversely Affect determination for the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS. Additionally, the Agency has determined that there is the potential for modification of the designated critical habitat for the BCB, DS, TG, CTS-CC, and CTS-SB from the use of the chemical. Given the LAA and potential modification of designated critical habitat determinations, a description of the baseline status and cumulative effects is provided in Attachment III.

A summary of the risk conclusions and effects determinations for the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS and their critical habitat, given the uncertainties discussed in Section 6 and Attachment I, is presented in **Table 1-1**. Use specific effects determinations are provided in **Table 1-2** and **Table 1-3**.

In this document, only currently marketed uses of PCNB on cole crops, cotton, potatoes, and turf were assessed. Since LAA determinations were made for all seven listed species evaluated, the inclusion of additional uses of PCNB are not expected to affect the overall outcome of this assessment. However, due to differences in exposure modeling and other analysis factors associated with individual agricultural and non-agricultural uses, it is not possible to predict which additional uses of PCNB (if marketed) would result in exceedances of Agency LOCs for direct and indirect effects to taxa linked to the species in this assessment.

Table 1-1. Effects Determination Summary for Effects of PCNB on the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS

Species	Effects Determination	Basis for Determination
Bay Checkerspot Butterfly (*Euphydryas editha bayensis*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects** ***Terrestrial*** Acute and chronic RQ values could not be calculated for direct effects to the BCB because there is no definitive acute toxicity endpoint for the honeybee (surrogate for BCB in this assessment) and no relevant chronic terrestrial insect data from the open literature. If EECs for arthropods were compared to the highest dose tested in the acute contact honeybee toxicity study (100 µg/bee converted to 781.25 µg/g based on assumed weight of 0.128 g/bee), the ratio would range from 0.24 to 6.3 for the various PCNB uses, indicating that there is some uncertainty surrounding risk to the BCB based on lack of definitive surrogate toxicity data. In order for there not to be risks of concern to the BCB, the definitive honeybee LD_{50} would have to be ≥12,503 µg/bee to not exceed the terrestrial invertebrate listed species LOC of 0.05; this is approximately one-tenth the weight of an adult honeybee. Therefore, due to the relatively high application rates of PCNB, there is significant uncertainty in the potential for effects to the BCB based on available toxicity information. Given the uncertainty in the risk to the BCB based on available honeybee data, combined with the lack of toxicity data on the effects of PCNB on lepidopterans, there is insufficient effects information to determine potential direct effects to the BCB for all PCNB uses evaluated in this assessment. **Potential for Indirect Effects** ***Terrestrial food items, habitat*** The BCB relies on terrestrial plants exclusively for both food and habitat and has an obligate relationship with dicots. However, no relevant terrestrial plant toxicity data have been identified for PCNB or its degradates. The only two incidents reported for PCNB affected terrestrial plants. Due to the lack of effects data on terrestrial plants, there is insufficient information to determine potential indirect effects to the BCB for all PCNB uses evaluated in this assessment.
California Tiger Salamander (All 3 DPS) (*Ambystoma californiense)*	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects** ***Aquatic-phase (Eggs, Larvae, and Adults)*** Acute RQ values based on freshwater fish (surrogate for aquatic-phase amphibians) toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the CTS for those uses. In addition, individual effect probabilities for CTS based on freshwater fish data range from 1-in-9.1 to 1-in-2.8×10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the aquatic-phase CTS as a result of cole crop and turf uses of PCNB. ***Terrestrial-phase (Juveniles and Adults)*** The chronic risk LOC for birds (surrogate for terrestrial-phase CTS) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, when worms are considered as a dietary item for the CTS, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CTS, indicating the potential for chronic risk to the CTS for these crop uses of PCNB based on worm dietary items.

Species	Effects Determination	Basis for Determination
		Acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (surrogate for terrestrial-phase amphibians). Therefore, there is a potential for direct effects to the terrestrial-phase CTS for all PCNB uses evaluated in this assessment. **Potential for Indirect Effects** ***Aquatic prey items, aquatic habitat, cover, and primary productivity*** RQs for freshwater fish (surrogate for aquatic-phase amphibian dietary items) only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the CTS based on this prey component (*i.e.*, non-listed aquatic-phase amphibians) is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-1x10^{11} across PCNB uses. Therefore, there is low potential for indirect effects to the CTS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. RQs cannot be calculated for aquatic plants due to a lack of toxicity data, representing a source of uncertainty. These results suggest that there is little potential for any of the uses of PCNB to affect the prey-base of the aquatic-phase CTS, but effects on its habitat are uncertain due to lack of toxicity data on aquatic plants. ***Terrestrial prey items, habitat*** As described for the BCB above, there is uncertainty in potential effects to terrestrial invertebrates (prey item) for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB. Based on submitted rat two-generation reproductive toxicity data, the chronic risk LOC for mammals is exceeded for all PCNB uses, indicating that small mammal prey may be adversely affected by PCNB at current use rates. EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammals. Therefore, there is the potential for indirect effects to CTS habitat based on chronic risk to small mammals. Based on these results, there is the potential for all uses of PCNB evaluated in this assessment to indirectly affect the terrestrial-phase CTS via terrestrial invertebrate prey as well as habitat in small mammal burrows.
California Clapper Rail (*Rallus longirostris obsoletus*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects** ***Terrestrial*** Based on avian reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses except for cotton. EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the CCR. In addition, when worms are considered as a dietary item for the CCR, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CCR, indicating the potential for chronic risk to the CCR for these crop uses of PCNB based on worm

Species	Effects Determination	Basis for Determination
		dietary items. Acute effects cannot be precluded for any PCNB uses because there is a non-definitive endpoint for acute toxicity to birds. Therefore, there is the potential for direct effects to the CCR for all PCNB uses evaluated in this assessment.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover, and primary productivity Freshwater fish RQs only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the CCR for all PCNB uses based on this prey component is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-1x10^{11} across PCNB uses. Therefore, there is low potential for indirect effects to the CCR based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. No acute or chronic RQ values representing any uses of PCNB exceed LOCs for estuarine/marine fish. The probability of an individual effect for estuarine/marine fish ranges from 1-in-9.3x10^4 to 1-in-3.5x10^6 across PCNB uses. Therefore, there is low potential for indirect effects to the CCR based on this prey component. Estuarine/marine invertebrate acute RQ values exceed the non-listed species LOC (0.5) for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, indirect effects to the CCR are anticipated based on this prey component. These results show that PCNB is likely to indirectly affect the CCR via freshwater fish and estuarine/marine invertebrate prey under exposure scenarios evaluated in this assessment.
		Terrestrial prey items, riparian habitat As describe for direct effects to the CCR above, there is the potential for chronic effect to birds (as prey for CCR) for all PCNB uses except for cotton. In addition, the possibility of acute affects to birds cannot be precluded for any of the PCNB uses based on available data. As described for the indirect effects to the CTS above, the mammalian chronic risk LOC is exceeded for all PCNB uses, indicating that there is potential for indirect effects to the CCR based on potential effects to small mammal prey items. As described for the BCB above, there is uncertainty in potential effects to terrestrial invertebrates for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB. No relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the

Species	Effects Determination	Basis for Determination
		CCR based on this prey item. Based on these results, there is the potential for all uses of PCNB to indirectly affect the CCR via effects on bird, mammal, and possibly invertebrate prey.
California Freshwater Shrimp (*Syncaris pacifica*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Aquatic Acute RQs do not exceed the acute risk to non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-$1x10^{11}$ to 1-in-$1x10^{16}$ across PCNB uses. Therefore, direct effects are not expected to the CFWS based on risk to freshwater invertebrate prey under the exposure scenarios evaluated in this assessment and based on available toxicity data. It should be noted that there is significant uncertainty associated with this conclusion because of the higher toxicity of PCNB to mysid shrimp (estuarine/marine crustacean) compared to *Daphnia magna* (freshwater crustacean). Mysid shrimp are more closely related to the CFWS, and may indicate that effects to the CFWS may be underestimated based on the toxicity endpoint from *D. magna*. In addition, open-literature studies on freshwater benthic invertebrate species with pentachlorobenzene (a degradate of PCNB) reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs for freshwater invertebrates in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species. Moreover, if any of the other freshwater invertebrate species were used to calculate RQs for the CFWS, it would result in exceedance of the acute risk to listed species LOC. Based on toxicity data from freshwater invertebrates, there is low potential for direct effects to the CFWS as a result of PCNB uses. However, there is uncertainty associated with this conclusion based on higher toxicity of PCNB to mysid shrimp and several freshwater invertebrates and the potential for bioaccumulation of PCNB in CFWS aquatic dietary items.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover, and primary productivity As described for the direct effects to the CFWS above, there is low potential for direct effects to freshwater inverterbrates as a result of PCNB uses. However, there is uncertainty associated with this conclusion based on higher toxicity of PCNB to mysid shrimp and several freshwater invertebrates and the potential for bioaccumulation of PCNB in CFWS aquatic dietary items. RQs cannot be calculated for aquatic plants due to a lack of toxicity data, representing a source of uncertainty concerning the effects of PCNB on plant food sources and habitat. These results suggest there is low potential for any uses of PCNB to indirectly impact the freshwater invertebrate prey-base of the CFWS. However, there is significant uncertainty associated with this finding due to lack of toxicity data on aquatic plants and the potential underestimation of toxicity for aquatic invertebrates.

Species	Effects Determination	Basis for Determination
		Terrestrial prey items, riparian habitat No relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is uncertain whether PCNB is likely to indirectly affect the CFWS via terrestrial plant food and habitat.
San Francisco Garter Snake (*Thamnophis sirtalis tetrataenia*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects** ***Terrestrial*** The chronic risk LOC for birds (as a surrogate for SFGS) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, when worms are considered as a dietary item for the SFGS, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the SFGS, indicating the potential for chronic risk to the SFGS for these crop uses of PCNB based on worm dietary items. Acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (surrogate for SFS). Therefore, there is a potential for direct effects to the SFGS for all PCNB uses evaluated in this assessment. **Potential for Indirect Effects** ***Aquatic prey items, aquatic habitat, cover, and primary productivity*** Freshwater fish (and aquatic-phase amphibian) RQs only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the SFGS for all PCNB uses based on this prey component is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the SFGS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. These results suggest that there is low potential for any PCNB use to indirectly affect the SFGS via freshwater fish and amphibian prey under exposure scenarios evaluated in this assessment. ***Terrestrial prey items, riparian habitat*** The chronic risk LOC for birds (as a surrogate for terrestrial-phase amphibians and reptiles) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (as a surrogate for terrestrial-phase amphibians and reptiles). Therefore, there is a potential for indirect effects to the SFGS for all PCNB uses evaluated in this assessment. As described for the indirect effects to the CTS above, the mammalian chronic risk LOC is exceeded for all PCNB uses, indicating that there is potential for indirect effects to the SFGS based on potential effects to small mammal prey items. As described for the BCB above, there is uncertainty in potential effects to

<table>
<tr><th>Species</th><th>Effects Determination</th><th>Basis for Determination</th></tr>
<tr><td></td><td></td><td>terrestrial invertebrates (prey item) for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB.

Based on these results, there is the potential for PCNB to indirectly affect the SFGS via reptile, amphibian, mammal, and possibly invertebrate prey as well as the mammal burrow component of habitat.</td></tr>
<tr><td rowspan="4">Delta Smelt (Hypomesus transpacificus)</td><td rowspan="4">May Affect, Likely to Adversely Affect (LAA)

[Applies to cole crop, cotton, potato, and turf uses of PCNB]</td><td>Potential for Direct Effects</td></tr>
<tr><td>Aquatic
Acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the DS for those uses. In addition, individual effect probabilities for DS based on freshwater fish data range from 1-in-9.1 to 1-in-2.8×10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the DS as a result of cole crop and turf uses of PCNB.</td></tr>
<tr><td>Potential for Indirect Effects</td></tr>
<tr><td>Aquatic prey items, aquatic habitat
For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-1×10^{11} across PCNB uses. Therefore, there is low potential for indirect effects to the DS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment.

Conversely, for estuarine/marine invertebrates, acute RQ values exceed the non-listed species LOC (0.5) for estuarine/marine invertebrates for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, there is the potential for indirect effects to the DS based on this prey component.

RQs cannot be calculated for terrestrial or aquatic plants due to a lack of toxicity data, representing a source of uncertainty.

Based on these results, PCNB is likely to indirectly affect the DS via estuarine/marine invertebrate prey, and possibly terrestrial and aquatic plant habitat, under exposure scenarios evaluated in this assessment.</td></tr>
<tr><td rowspan="4">Tidewater Goby (Eucyclogobius newberryi)</td><td rowspan="4">May Affect, Likely to Adversely Affect (LAA)

[Applies to cole crop, cotton, potato, and turf uses of PCNB]</td><td>Potential for Direct Effects</td></tr>
<tr><td>Aquatic
Acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the TG for those uses. In addition, individual effect probabilities for TG based on freshwater fish data range from 1-in-9.1 to 1-in-2.8×10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the DS as a result of cole crop and turf uses of PCNB.</td></tr>
<tr><td>Potential for Indirect Effects</td></tr>
<tr><td>aquatic prey items, terrestrial/aquatic habitat</td></tr>
</table>

Species	Effects Determination	Basis for Determination
		For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the TG based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. Conversely, for estuarine/marine invertebrates, acute RQ values exceed the non-listed species LOC (0.5) for estuarine/marine invertebrates for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, there is the potential for indirect effects to the TG based on this prey component. RQs cannot be calculated for terrestrial or aquatic plants due to a lack of toxicity data, representing a source of uncertainty. Based on these results, PCNB is likely to indirectly affect the TG via estuarine/marine invertebrate prey, and possibly terrestrial and aquatic plant habitat, under exposure scenarios evaluated in this assessment.

Table 1-2. Use Specific Summary of the Potential for Adverse Effects to Aquatic Taxa

Uses	Potential for Effects to Identified Taxa Found in the Aquatic Environment:									
	Estuarine/Marine Vertebrates[1]		Tidewater Goby, Delta Smelt, California Tiger Salamander (all DPS) and Freshwater Vertebrates[2]		California Freshwater Shrimp and Freshwater Invertebrates[3]		Estuarine/Marine Invertebrates[4]		Vascular Plants[5]	Non-vascular Plants[5]
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic		
Cole crops	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC
Cotton	No	No	No	No	No	No	No	Yes	UNC	UNC
Potatoes	No	No	No	No	No	No	No	Yes	UNC	UNC
Turf (foliar)	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC
Turf (granular)	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC

UNC = uncertain due to lack of effects data or non-definitive toxicity data where risks of concern cannot be precluded
DPS = distinct population segments
[1] A yes in this column indicates a potential for indirect effects to CCR.
[2] A yes in this column indicates a potential for direct effects to the TG, DS, CTS-CC, CTS-SC, and CTS-SB, and potential for indirect effects to SFGS, CCR, CTS-CC, CTS-SC, and CTS-SB.
[3] A yes in this column indicates a potential for direct effects to the CFWS and indirect effects to the CFWS, TG, DS, SFGS, CCR, CTS-CC, CTS-SB, CTS-SC.
[4] A yes in this column indicates a potential for indirect effects to TG, DS, CCR.
[5] A yes in this column indicates a potential for indirect effects to SFGS, CCR, TG, DS, CTS-CC, CTS-SC, CTS-SB, and CFWS.
* RQ exceeds the LOC for listed species (potential for direct effects) but not for non-listed species (no potential for indirect effects).

Table 1-3. Use Specific Summary of the Potential for Adverse Effects to Terrestrial Taxa

Uses	Potential for Effects to Identified Taxa Found in the Terrestrial Environment:										
	Small Mammals[1]		California Clapper Rail and Small Birds[2]		California Tiger Salamander (all DPS) and Amphibians[3]		San Francisco Garter Snake and Reptiles[4]		Bay Checkerspot Butterfly and Terrestrial Invertebrates (Acute)[5]	Dicots[6]	Monocots[6]
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic			
Cole crops	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Cotton	No	Yes	UNC	No	UNC	No	UNC	No	UNC*	UNC	UNC
Potatoes	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Turf (foliar)	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Turf (granular)	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC

UNC = uncertain due to lack of effects data or non-definitive toxicity data where risks of concern cannot be precluded
DPS = distinct population segments
[1] A yes in this column indicates a potential for indirect effects to SFGS, CCR, CTS-CC, CTS-SC, CTS, and CTS-SB.
[2] A yes in this column indicates a potential for direct effects to CCR and indirect effects to the CCR, SFGS, CTS-CC, CTS-SC, and CTS-SB.

[3] A yes in this column indicates a potential for direct CTS-CC, CTS-SC, CTS-SB, and indirect effects to CTS-CC, CTS-SC, CTS-SB, SFGS, and CCR.
[4] A yes in this column indicates the potential for direct and indirect effects to SFGS and other reptiles.
[5] A yes in this column indicates a potential for direct effect to BCB and indirect effects to SFGS, CCR, CTS-CC, CTS-SC, and CTS-SB.
[6] A yes in this column indicates a potential for indirect effects to BCB, SFGS, CCR, TG, DS, CTS-CC, CTS-SC, CTS-SB, and CFWS.
* There is some indication that PCNB use on cole crops could adversely impact earthworms resulting in indirect effects to the CTS, CCR, and SFGS, which could prey on worms (see Section 5.2.2.b for details).

Based on the conclusions of this assessment, a formal consultation with the U. S. Fish and Wildlife Service under Section 7 of the Endangered Species Act should be initiated.

When evaluating the significance of this risk assessment's direct/indirect and adverse habitat modification effects determinations, it is important to note that pesticide exposures and predicted risks to the assessed listed species and their resources (*i.e.*, food and habitat) are not expected to be uniform across the action area. In fact, given the assumptions of drift and downstream transport (*i.e.*, attenuation with distance), pesticide exposure and associated risks to the species and its resources are expected to decrease with increasing distance away from the treated field or site of application. Evaluation of the implication of this non-uniform distribution of risk to the species would require information and assessment techniques that are not currently available. Examples of such information and methodology required for this type of analysis would include the following:

- Enhanced information on the density and distribution of BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG life stages within the action area and/or applicable designated critical habitat. This information would allow for quantitative extrapolation of the present risk assessment's predictions of individual effects to the proportion of the population extant within geographical areas where those effects are predicted. Furthermore, such population information would allow for a more comprehensive evaluation of the significance of potential resource impairment to individuals of the assessed species.
- Quantitative information on prey base requirements for the assessed species. While existing information provides a preliminary picture of the types of food sources utilized by the assessed species, it does not establish minimal requirements to sustain healthy individuals at varying life stages. Such information could be used to establish biologically relevant thresholds of effects on the prey base, and ultimately establish geographical limits to those effects. This information could be used together with the density data discussed above to characterize the likelihood of adverse effects to individuals.
- Information on population responses of prey base organisms to the pesticide. Currently, methodologies are limited to predicting exposures and likely levels of direct mortality, growth or reproductive impairment immediately following exposure to the pesticide. The degree to which repeated exposure events and the inherent demographic characteristics of the prey population play into the extent to which prey resources may recover is not predictable. An enhanced understanding of long-term prey responses to pesticide exposure would allow for a more refined determination of the magnitude and duration of resource impairment, and together with the information described above, a more complete prediction of effects to individual species and potential modification to critical habitat.

2. Problem Formulation

Problem formulation provides a strategic framework for the risk assessment. By identifying the important components of the problem, it focuses the assessment on the most relevant life history stages, habitat components, chemical properties, exposure routes, and endpoints. The structure of this risk assessment is based on guidance contained in U.S. EPA's *Guidance for Ecological Risk Assessment* (USEPA, 1998), the Services' *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998) and is consistent with procedures and methodology outlined in the Overview Document (USEPA, 2004) and reviewed by the U.S. Fish and Wildlife Service and National Marine Fisheries Service (USFWS/NMFS/NOAA, 2004).

2.1. Purpose

The purpose of this endangered species assessment is to evaluate potential direct and indirect effects on individuals of the federally threatened BCB, CTS-CC, and DS, and federally endangered CCR, CFWS, CTS-SC, CTS-SB, SFGS, and TG arising from FIFRA regulatory actions regarding cole crops, golf course GTF, cotton, potatoes and ornamental bulbs (dip/soak only) uses of PCNB. This ecological risk assessment has been prepared consistent with a stipulated injunction in the case *Center for Biological Diversity (CBD) vs. EPA et al.,* (Case No. 07-2794-JCS) entered in Federal District Court for the Northern District of California on May 17, 2010.

In this assessment, direct and indirect effects to the BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and potential modification to designated critical habitat for the BCB, CTS-CC, CTS-SB, DS, and TG are evaluated in accordance with the methods described in the Agency's Overview Document (USEPA, 2004).

The BCB was listed as threatened in 1987 by the USFWS. The species primarily inhabits native grasslands on serpentine outcrops around the San Francisco Bay Area in California. The PCEs for BCBs are areas on serpentinite-derived soils that support the primary larval host plant (*i.e.*, dwarf plantain) and at least one of the species' secondary host plants. Additional BCB PCEs include the presence of adult nectar sources, aquatic features that provide moisture during the spring drought and areas that provide adequate shelter during the summer diapause.

There are currently three CTS Distinct Population Segments (DPSs): the Sonoma County (SC) DPS, the Santa Barbara (SB) DPS, and the Central California (CC) DPS. Each DPS is considered separately in the risk assessment as they occupy different geographic areas. The main difference in the assessment will be in the spatial analysis. The CTS-SB was listed by the USFWS as endangered in 2000, the CTS-SC in 2002, and the CTS-CC as threatened in 2004. The CTS-SB and CTS-SC were downlisted from endangered to threatened in 2004 by the USFWS, however, the downlisting was vacated by the U.S. District Court. Therefore, the Sonoma and Santa Barbara DPSs are currently listed as endangered while the CTS-CC is listed as threatened. All CTS populations utilize vernal pools, semi-permanent ponds, and permanent ponds, and the terrestrial environment in California. The aquatic environment is essential for breeding and reproduction and mammal burrows are also important habitat for aestivation. The PCEs for the CTS are standing bodies of freshwater sufficient for the species to complete the aquatic portion of its life cycle that are adjacent to barrier-free uplands that contain small mammal burrows. An additional PCE is upland areas between sites (as described above) that allow for dispersal of the species.

The DS was listed as threatened on March 5, 1993 (58 FR 12854) by the USFWS (USFWS, 2007). DS are mainly found in the Suisun Bay and the Sacramento-San Joaquin estuary near San Francisco Bay. During spawning DS move into freshwater. The PCEs for DSs are shallow fresh or brackish backwater sloughs for egg hatching and larval viability, suitable water with adequate river flow for larval and juvenile transport, suitable rearing habitat, and unrestricted access to suitable spawning habitat.

The CCR was listed by the USFWS as an endangered species in 1970. The species is found only in California in coastal wetlands along the San Francisco estuary and Suisun Bay.

The CFWS was listed as endangered in 1988 by the USFWS. The CFWS inhabits freshwater streams in Central California in the lower Russian River drainage and westward to the Pacific Ocean and coastal streams draining into Tomales Bay and southward into the San Pablo Bay.

The SFGS was listed as endangered in 1967 by the USFWS. The species is endemic to the San Francisco Peninsula and San Mateo County in California in densely vegetated areas near marshes and standing open water.

The TG was listed as endangered in 1994 by the USFWS. The range of the TG is limited to coastal brackish water habitats along the coast of California. The PCEs for TGs are persistent, shallow aquatic habitats with salinity from 0.5 parts per thousand (ppt) to 12 ppt, that contain substrates suitable for the construction of burrows and submerged aquatic plants that provide protection. An additional PCE is the presence of sandbars that at least partially closes a lagoon or estuary during the late spring, summer, and fall.

In accordance with the Overview Document, provisions of the ESA, and the Services' *Endangered Species Consultation Handbook,* the assessment of effects associated with registrations of PCNB is based on an action area. The action area is the area directly or indirectly affected by the federal action, as indicated by the exceedance of the Agency's Levels of Concern (LOCs). It is acknowledged that the action area for a national-level FIFRA regulatory decision associated with a use of PCNB may potentially involve numerous areas throughout the United States and its Territories. However, for the purposes of this assessment, attention will be focused on relevant sections of the action area including those geographic areas associated with locations of the BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and their designated critical habitat within the state of California. As part of the "effects determination," one of the following three conclusions will be reached separately for each of the assessed species in the lawsuits regarding the potential use of PCNB in accordance with current labels:

- "No effect";
- "May affect, but not likely to adversely affect"; or
- "May affect and likely to adversely affect".

Additionally, for habitat and PCEs, a "No Effect" or a "Habitat Modification" determination is made.

A description of routine procedures for evaluating risk to the San Francisco Bay Species is provided in Attachment I.

2.2. Scope

The end result of the EPA pesticide registration process (*i.e.*, the FIFRA regulatory action) is an approved product label. The label is a legal document that stipulates how and where a given pesticide may be used. Product labels (also known as end-use labels) describe the formulation type (*e.g.*, liquid or granular), acceptable methods of application, approved use sites, and any restrictions on how applications may be conducted. Thus, the use or potential use of PCNB in accordance with the approved product labels for California is "the action" relevant to this ecological risk assessment.

PCNB is a fungicide that is currently marketed for use on cole crops (broccoli, Brussels sprouts, cabbage, cauliflower), golf course turf (only greens, tees, fairways; GTF), cotton, potatoes, and ornamental bulbs (dip/soak only), and collards, kale and mustard greens (in GA only). Formulation types currently approved and marketed include a flowable (Turfcide® 4F; EPA Reg. #5481-8992) and a granular (Turfcide 10% Granular®; EPA Reg. #5481-8988). Only the following currently marketed and approved uses are permitted in CA and are considered as part of the federal action evaluated in this assessment: cole crops, golf course GTF, cotton, potatoes and ornamental bulbs (dip/soak only).

Although current registrations of PCNB allow for use nationwide, this ecological risk assessment and effects determination addresses currently registered uses of PCNB in portions of the action area that are reasonably assumed to be biologically relevant to the assessed species and their designated critical habitat. Further discussion of the action area for the assessed species and their critical habitat is provided in Section 2.7.

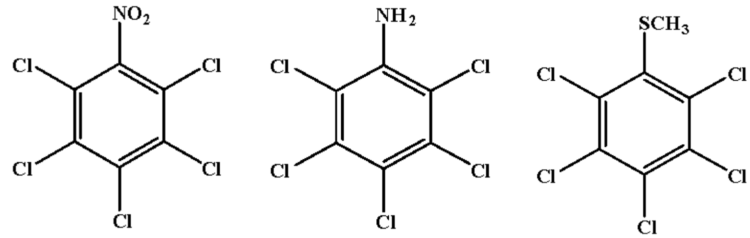
2.2.1. Evaluation of Degradates

PCNB has several environmental degradates and contaminants (present in technical PCNB) that are of toxicological concern. Degradates and contaminants are of toxicological concern when their structure includes a polychlorinated phenyl ring.

Environmental degradates and/or impurities of PCNB that were identified in the environmental fate and ecological risk assessment chapter for the Registration Eligibility Decision (RED) for PCNB (USEPA, 2005) include:

- pentachloroaniline (PCA),
- pentachlorothioanisole (PCTA),
- pentachlorobenzene (PCB; also present as a contaminant),
- pentachlorophenol (PCP),
- pentachlorothioanisole sulfoxide (PCTASO), and
- pentachlorothioanisole sulfone (PCTASO$_2$)
- hexachlorobenzene (HCB; present as a contaminant).

Structures for PCNB and its two main degradates are presented in **Figure 2-1**.



PCNB PCA PCTA

Figure 2-1. Structures of PCNB and its two main environmental degradates.

Technical PCNB contains several manufacturing impurities, which are termed "contaminants" in this assessment. In EFED's environmental fate and ecological risk assessment RED chapter for PCNB, hexachlorobenzene (HCB) and PCB were identified as contaminants (USEPA, 2005). For HCB and PCB, EPA allows a maximum of 500 ppm and 100 ppm, respectively, in technical PCNB.

According to samples analyzed by the Biologic and Economic Analysis Division (BEAD) labs as well as data reported in the Canadian Reevaluation Decision for the formulated product Quintozene (*i.e.,* PCNB; Health Canada 2009), polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are also present in technical PCNB.

Although not considered in EFED's 2005 ecological risk assessment chapter for the PCNB RED, PCDDs and PCDFs present in PCNB formulations are considered to be of concern and are included in the current assessment to a limited extent. They are not reflected in half-lives from submitted studies, as they were not monitored in the submitted laboratory or field studies. However, these chemicals are considered to be persistent, bioaccumulative, and are not readily metabolized by mammals (Van Den Berg *et al.,* 2006). In general, considering the logarithm of the octanol-water partition coefficient (log K_{ow}) values of PCDDs and PCDFs (range of 7.9-9.5) relative to those of PCNB and its degradates (log K_{ow}=5.0), these contaminants would be expected to bioaccumulate to a greater extent than PCNB and its degradates.

2.2.2. Evaluation of Mixtures

The Agency does not routinely evaluate mixtures of active ingredients in its risk assessments, including mixtures of multiple active ingredients in product formulations or those in the applicator's tank. In the case of the product formulations of active ingredients (*i.e.*, a registered product containing more than one active ingredient), each active ingredient is subject to an individual risk assessment for regulatory decision regarding the active ingredient on a particular use site. If effects data are available for a formulated product containing more than one active ingredient, they may be used qualitatively or quantitatively in accordance with the Agency's Overview Document and the Services' Evaluation Memorandum (USEPA, 2004; USFWS/NMFS/NOAA, 2004).

PCNB does not have registered products that contain multiple active ingredients.

2.3. Previous Assessments

The environmental fate and ecological risk assessment written in support of the Reregistration Eligibility Decision (RED) for PCNB was finalized in 2005, and the RED was completed and signed in 2006 (USEPA, 2006). In the ecological risk assessment, EFED concluded that for aquatic animals, exposure was likely and acute risk LOCs were exceeded. Chronic risk LOCs were also exceeded for freshwater fish and invertebrates; chronic effects include reduced number of eggs produced and reductions in the number of young surviving. The likelihood of chronic risk to estuarine/marine animals was uncertain since no data were available with which to assess the chronic toxicity of PCNB to estuarine/marine animals. However, based on acute-to-chronic ratios derived from freshwater studies, estimated NOAECs for estuarine/marine invertebrates suggested that chronic risk was possible.

At the time of the 2005 assessment, data were not available on the effects of PCNB on either aquatic or terrestrial plants; therefore, it was not possible to assess potential risks to plants. At that time, open literature suggested that, like aquatic animals, aquatic plants are able to accumulate PCNB residues and serve as a route of entry for PCNB into the aquatic food chain, although compounds such as PCNB and PCA are associated with relatively low dietary absorption efficiency and short metabolic half-lives in some aquatic animals. However, the likely persistence and toxicity of PCNB and its degradates coupled with the chemicals' propensity to bioconcentrate, were concerns that underscore the potential vulnerability of aquatic communities to PCNB exposure resulting from labeled uses of the fungicide.

In the 2005 assessment, EFED concluded that given the persistence of PCNB and its degradates in the aquatic environment and the propensity of the compounds to bioconcentrate, its presence in the benthic sediments was expected to serve as a likely route of exposure to benthic fauna and as a means of entry into aquatic food chains where the compound and/or its degradates may biomagnify as well. The available data indicated that residues of PCNB and its degradates had been detected in aquatic plants and invertebrates; PCNB residues had also been detected in bottom-feeding fish.

Although screening-level models indicated that terrestrial exposure to PCNB was likely through consumption of treated seed and foliar residues, the likelihood of acute mortality (risk) to either birds or mammals, based on a deterministic (risk quotient) approach, was determined to be low given that PCNB is practically nontoxic to both birds and mammals on an acute exposure basis. However, chronic risk LOCs were exceeded for birds and mammals following consumption of PCNB-treated seed and PCNB residues on other treated forage items. EFED concluded that chronic effects such as reductions in the number of eggs laid, number of viable embryos and numbers of 14-day survivors may result from chronic exposure of birds to PCNB. Chronic exposure to PCNB during a rat 2-generation reproduction study resulted in reduced weight in both offspring and adult animals.

EFED also noted in the 2005 document that PCNB meets established criteria for classifying it as a persistent, bioaccumulative, toxic (PBT) chemical. However, the extent to which the compound would bioaccumulate in aquatic environments was uncertain. It was concluded that the extent to which PCNB and its degradates bioconcentrate and/or bioaccumulate in the aquatic food chain will only accentuate the other ecological risks documented in the assessment and that potential chronic risks to terrestrial animals from exposure to PCNB through the food chain will depend on the extent to which terrestrial animals rely on aquatic organisms in their diet.

In March 2010, EFED finalized the document "*2008 Science Advisory Panel Meeting Follow Up: Assessment of the Bioaccumulation and Long-Range Transport Potential (LRTP) and of Pentachloronitrobenzene (PCNB) and Associated Ecological Risks*" (USEPA, 2010) after considering the Agency's Science Advisory Panel (SAP) recommendations on assessing compounds with PBT-like properties. The assessment focused on bioaccumulation and long-range transport potential of PCNB and several of its degradates and contaminants/impurities. The assessment used a refined approach relative to the 2005 RED, based on SAP recommendations. The Agency concluded that

"*PCNB and its degradates and contaminants are considered sufficiently persistent and likely to bioaccumulate in aquatic food chains to represent a chronic risk to mammals that consume aquatic organisms. In addition, PCNB is expected to move through long-range atmospheric transport to considerable distances from its site of application. Technical PCNB contains several contaminants, including hexachlorobenzene, polychlorinated dibenzodioxins and polychlorinated dibenzofurans, which are listed under the Stockholm Convention as Persistent Organic Pollutants (POPs). The weight of evidence suggests that PCNB, when considered with its residues and contaminants of concern, is a persistent, bioaccumulative, toxic chemical (PBT).*"

2.4. Environmental Fate Properties

Table 2-1 lists the physical-chemical properties of PCNB, a moderately volatile, relatively high molecular weight, neutral organic compound with low water solubility. **Table 2-2** lists the other environmental fate properties of PCNB, along with the major and minor degradates detected in the submitted environmental fate and transport studies. Based on the submitted environmental fate data, its physical-chemical properties, use patterns, and information found in the published literature, PCNB and it residues of concern are, in general, expected to be persistent in the environment. Parent PCNB and the major degradate PCA are slightly mobile to hardly mobile in most soils, while the degradate PCTA is hardly mobile (FAO classification scheme).

Table 2-1. Physical-chemical Properties of PCNB

Property	**Parent Compound**	
	Value and units	**MRID or Source**
Molecular Weight	295.3 g/mole	USEPA, 2005
Chemical Formula	C6-Cl5-N-O2	Merck Index (13th Ed., 2001)
Density/ Relative Density/ Bulk Density	1.718 g/cm^3	Merck Index (13th Ed., 2001)
Vapor Pressure	1.13 x 10^{-4} Torr @ 25°C	
Henry's Law Constant	9.980E-005 atm-m^3/mole @ 25°C	Estimated from water solubility and vapor pressure

Property	Parent Compound	
	Value and units	MRID or Source
Water Solubility	0.44 mg/L @ 25°C	
Octanol – water partition coefficient (Log K_{OW})	5.0 @ 25°C	USEPA, 2005.
Dissociation Constant (pK_a and/or pK_b)	Not Applicable	PCNB is a neutral organic
Air-water partition coefficient (log K_{AW})	-2.743	EPISuite experimental database.
Octanol-air partition coefficient (log K_{OA})	7.38	Estimated using EPISuite
UV/visible light absorption	301 nm	Khan *et al., 2011*

Table 2-2. Summary of PCNB Environmental Fate Properties for the Parent Compound and Degradates (Where Available)

Study	Value and unit	Major Degradate (max) *Minor Degradates (max)*	MRID # or Citation	Study Classification, Comment
Abiotic Hydrolysis	Stable (pH 5, 7, 9; 25 °C)	None	40865301 40972601	Acceptable
Air Photolysis	No data	–	–	–
Atmospheric Degradation	No data Half-life = 1480 days, estimated			Estimated using EPIWIN v4.10
Direct Aqueous Photolysis	Half-life[1] = 2.5 days, 1.12 days, pH 5	chlorinated hydroxybenzenes and/or chloronitrophenols; unidentified degradates	42606201 42606202 42336201	Acceptable
Soil Photolysis	Stable		41004801 41713201	Acceptable
Aerobic Soil Metabolism	Half-life[1] = 77, 189 days, sandy loam (parent only) Half-life[1] = 983, 1052 days, sandy loam (total residues)	PCA (0.90 ppm) PCTA (0.81 ppm) PCB (0.27 ppm) PCTASO (0.37 ppm) $PCTASO_2$ (0.61ppm)	42911902 41384501 41713202 42112801	Acceptable Acceptable
Anaerobic Soil Metabolism	Half-life[2] = 9 days & <30 (DT_{50}) days, both in sandy loam (parent only) Half-life[1] = 334 days, sandy loam (total residues)	PCA (87% in soil;27% in water) *PCTA* (3.2% in soil) *PCP* (2% in soil)	41203602 42094401 41384301 41686001 41713202 42112802	Acceptable Half-lives of questionable value due to scarcity of data points. Acceptable
Aerobic Aquatic	No data	–	–	–

Study	Value and unit	Major Degradate (max) *Minor Degradates (max)*	MRID # or Citation	Study Classification, Comment
Metabolism				
Anaerobic Aquatic Metabolism	No data	–	–	–
Freundlich solid-water distribution coefficient (K_F)	Parent PCNB K_F, 1/n 7.3 L/kg, 0.90, sand 15.5 L/kg, 0.87, sandy loam 19.1 L/kg, 0.83, silt loam 210 L/kg, 1.1, clay		41648201	Acceptable
Organic-carbon normalized distribution coefficient (K_{OC})	K_{OC} = 1588 L/kg silt loam; 2912 L/kg sand; 3870 L/kg sandy loam; 17508 L/kg clay		41648201	Acceptable
Freundlich solid-water distribution coefficient (K_F)	PCTA (degradate) 205 L/kg, sand 214 L/kg, sandy loam 503 L/kg silt loam 411 L/kg, silty clay loam		43500501	Acceptable
Organic-carbon normalized distribution coefficient (K_{OC})	PCTA (degradate) 18610 L/kg, sand 19395 L/kg, sandy loam 45662 L/kg silt loam 37267 L/kg, silty clay loam		43500501	Acceptable
Freundlich solid-water distribution coefficient (K_F)	PCA (degradate) 37 L/kg, sand 78 L/kg, sandy loam 39 L/kg silt loam 117 L/kg, silty clay loam		43500502	Acceptable
Organic-carbon normalized distribution coefficient (K_{OC})	PCA (degradate) 3337 L/kg, sand 7116 L/kg, sandy loam 3581 L/kg silt loam 10580 L/kg, silty clay loam		43500502	Acceptable
Volatility from Soil (Laboratory)	80% volatilized by 7 days, sandy loam		41178001 42507401	Acceptable Maximum volatilization within 24 hours of treatment.
Volatility from Soil (Field)	PCNB: 21.45 to 67.49 ng/liter (during four 2-hr samplings on day 0; silt loam) PCA: 3.32 to 9.61 ng/liter (during four 2-hr samplings on day 0; silt loam)		43751101	Acceptable Total of 65% of volatiles (PCNB & PCA) collected by day 1 post-application., with max. of each compound volatilized on day 0. Since >5 inches of rain fell in the

Study	Value and unit	Major Degradate (max) *Minor Degradates (max)*	MRID # or Citation	Study Classification, Comment
				study area (precluding air sampling on days 4 and 9) from days 3 to 9, the observed volatilization cannot be considered to be representative of that which might occur over time under more typical field conditions or in more arid regions.
Terrestrial Field Dissipation	Dissipation Half-life[1,2] = <u>Incorporated</u> 272 days, bareground, CA 459, bareground, GA 324, potatoes, MN 128, broccoli, CA 193, MN, potatoes <u>Non-incorporated</u> 57, bareground, GA 39, turf, CA	PCA (0.02 – 9.24 ppm) PCTA (0.008 – 0.73 ppm) PCB (0.02 - 0.24 ppm) PCTASO (0.07 ppm) $PCTASO_2$ (0.6 ppm)	41721401 43887401 43887402 43887403 43061501 41210501 & 42094403 & 42094404 41216401 & 42485901 41216402 & 42485902	Acceptable Supplemental Supplemental Acceptable Supplemental Acceptable Acceptable Acceptable
Bioconcentration Factor (BCF)- Bluegill Sunfish	Steady State BCF= 960-1100 L/kg wet wt whole fish 370-400 L/kg wet wt edible tissue 1800 L/kg wet wt nonedible tissue	N-acetyl S-pentachlorophenyl cystine (the cystinyl conjugate of PCNB): 0.8-1.2 ppb (edible); 25-31 ppb (nonedible) PCTA: 161-232 ppb (edible); 696-1095 ppb (nonedible)	40580202 41200001	Acceptable

Abbreviations: wt=weight

[1]Half-lives were calculated using the single-first order equation and nonlinear regression, unless otherwise specified.

[2]The value may reflect both dissipation and degradation processes.

2.4.1. Degradation and Dissipation in the Environment

The primary degradation pathway for PCNB in the environment may be aqueous photodegradation when the compound is present in an unsorbed state in clear and shallow surface water under favorable light conditions. Photodegradation of PCNB in water in the laboratory is moderately rapid, with half-lives on the order of a few days or less (MRID 42336201; MRIDs 42606201 and 42606202), although the photoproducts were not definitively identified in the submitted studies. However, direct photolytic degradation of PCNB in turbid and/or deeper waters may be limited by the attenuation of sunlight due to unfavorable conditions. Thus, caution must be used in extrapolating laboratory photolysis data to the environment. Also, adsorption of the compound to sediment (in runoff) or to suspended particles once it is in the water column (such as through drift) will decrease the amount of compound available for photolytic degradation.

PCNB is stable to hydrolysis and is effectively stable to photodegradation on soil (half-life of 80 days; MRIDs 41004801, 41713201). PCNB biodegrades slowly in aerobic soils, with half-lives of 77 and 189 days (MRIDs 42911902, 41384501, 41713202, and 42112801. Data from published literature also indicate that PCNB will be degraded slowly in aerobic soil, with half-lives of 4.7-9.7 months reported by Wang and Broadbent (1972) and half-lives of 213, 425, 535 and 699 days reported by Beck and Hansen (1974). While data in the submitted anaerobic soil metabolism studies were insufficient to calculate half-lives, PCNB is biodegraded more rapidly in anaerobic soil based on DT_{50}'s of <30 days (MRID's 41203602, 42094401). Although guideline study data on the aquatic metabolism of the compound were not submitted, data in the literature showed a rapid biodegradation of PCNB in anaerobic estuarine sediment (previously exposed to anthropogenic compounds from nearby industry), with a reported half-life of 0.8 days (Susarla *et al.,* 1996). Ko and Farley (1969) also reported rapid degradation in anaerobic (flooded) soil, with a reported half-life of approximately 2 weeks. Although PCNB may degrade rapidly in aquatic environments with anaerobic conditions, its major degradates, PCA and PCTA, may be more persistent in such systems. However, while Schauerte *et al.,* (1982) found a quantitative conversion of PCNB to the degradates PCA and PCTA in the sediment phase of an experimental pond study, Susarla *et al.,* (1996) found that in anaerobic sediment, PCNB degraded to multiple less-chlorinated anilines following the initial conversion to PCA. In the Schauerte *et al.,* study, the majority of PCNB residues remaining in the sediment at up to 145 weeks post-treatment were present in the 0- to 10-cm layer, but decreased with time. Overall, information on the rate and products of biodegradation in both aerobic and anaerobic aquatic systems is limited.

To assess PCNB and its residues of concern (PCA, PCTA, PCB, PCP, PCTASO, $PCTASO_2$ and HCB), soil metabolism half-lives were recalculated using concentration data (including volatiles) for use in determining estimated environmental concentrations (EECs). The PCDDs and PCDFs are not accounted for in the half-lives because they were not monitored in the submitted studies. The recalculated aerobic soil metabolism half-lives for total residues of concern are 983 days and 1,052 days. The recalculated value for the anaerobic soil metabolism half-life is 334 days. These values demonstrate the greater persistence of the total residues relative to the persistence of the parent compound alone.

In addition to degradation in the environment, PCNB can also be expected to dissipate from soil through volatilization. In submitted terrestrial field dissipation studies, PCNB dissipated more rapidly in the plots in which the pesticide was not incorporated, with a half-life range of 39-57 days (MRIDs 43061501; 43887403; 41210501, 42094403 and 42094404) versus a half-life range of 128-324 days (MRIDs 43887401; 43887402; 41721401; 41216401 and 42485901; 41216402 and 42485902) for plots with incorporated PCNB. Major degradates in those studies included PCA, PCTA and PCB. The manufacturing contaminant HCB was detected in all except one of the field studies, at maximum concentrations ranging from 0.03 to 0.076 ppm. In a study of field soils (cropped with potatoes) in Denmark to which PCNB had been applied over the previous 5 to 11 years, Beck and Hansen (1974) determined field half-lives ranging from 117 to 1059 days, with a mean of 434 days (for 22 values). Degradates observed in the Danish field soils were PCA, PCTA and PCB.

In a field volatility study, PCNB (as Terraclor® 75% WP) applied as a band application, at 10.2 lb ai/A, volatilized from an unvegetated silt loam soil in Mississippi; approximately 65% of the total volatiles trapped were collected during the first two days of the study. Volatilized PCNB was at a maximum on day 0 and declined throughout the rest of the 13-day study. The degradate PCA was also detected as a volatile. Total PCA volatilized was at a maximum on day 0 and declined throughout the rest of the study. It is noted, however, that the observed volatilization in the study may underestimate that which might have occurred in the field under different weather conditions. As over five inches of rain fell in the study area (precluding air sampling on days 4 and 9) from days 3 to 9, the observed volatilization may not be representative of that which might occur over time under more typical field conditions or in more arid regions. In a laboratory volatility study on sandy loam soil, approximately 80% of the applied PCNB was volatilized (as parent only) by day 7 (MRID 41178001. In a second laboratory volatility study, maximum volatilization occurred within 24 hours of treatment. The available data suggest that depending on use conditions, PCNB and some of its degradates may volatilize and contribute to atmospheric loading of the chemical(s). The extent to which this occurs cannot be quantified at this time; however, this loading would likely serve as a means through which PCNB could undergo long-range atmospheric transport.

2.4.2. Mobility

PCNB is slightly mobile to hardly mobile in most soils based on organic carbon partition coefficients (K_{oc}'s) of 1588-17,508 and using the mobility classification scheme of the Food and Agriculture Organization (FAO). The adsorption of PCNB correlates well with soil organic carbon content. Data from batch equilibrium studies, when considered along with results from screening models and terrestrial field dissipation studies, indicate a low potential for leaching to groundwater. Because adsorption of the compound is related to soil organic carbon content, a slightly higher, though still low, potential for leaching to groundwater might exist for PCNB in soils which are relatively low in organic matter, as is often the case with coarse-textured soils. In an experimental small pond study conducted in Germany, PCNB did not leach below the 15- to 20-cm sediment layer and did not leach significantly into the adjacent soil (Schauerte *et al.,* 1982).

Based on submitted guideline study data, the major degradate pentachloroaniline (PCA) is expected to be slightly mobile to hardly mobile in soil (K_{oc} range of 3337-10,580; MRID 43500502) and pentachlorothioanisole (PCTA) is expected to be hardly mobile in soil (K_{oc} range of 18,610-45,662; MRID 43500501). Data reported in the literature indicates that PCB will have slight mobility in soil (Forst *et al.,* 1994) or will be immobile (Sabljic *et al.,* 1995). These data are in basic agreement with observations made in the submitted terrestrial field dissipation studies, in which PCNB generally did not leach below the 6- to 12-inch soil depth; and PCA, PCTA and PCB generally remained in the 0- to 6-inch soil depth, although PCA and PCB leached into the 6- to 12-inch soil depth at some sites.

2.4.3. Bioaccumulation

Based on the results of a guideline study (MRID 40580202), PCNB has a very high potential to bioaccumulate in fish. Residues in bluegill sunfish (*Lepomis macrohirus*) exposed to [^{14}C]PCNB at ≤1.0 ppb were bioaccumulated to a greater extent in the viscera, with bioconcentration factors (BCF) of 1800X in each of two studies, versus in edible tissue, with BCF's of 370X and 400X; in whole fish tissue, measured BCF values were 960X and 1100X. As reported in the literature, PCNB was bioaccumulated in rainbow trout (*Oncorhynchus mykiss*) exposed to PCNB at 0.69 ppm, with BCF's of 114-261X (Niimi *et al.,* 1989). A BCF value of 238X was reported for PCNB in topmouth gudgeon (*Pseudorasbora parv*a; Kanazawa, 1981). Reported BCF values for guppies (*Poecilia reticulate*) exposed to PCNB range from 363 to 1030X (DeWolf, 1992). Higher BCF values have been reported for aquatic plants. Korte *et al.,* (1978) reported BCF values of 14,000X and 20,000X for PCNB in algae; and Wang *et al.,* (1996) reported 4508X for PCNB in the green algae *Chlorella fusca*. However, the values reported by Korte *et al.,* were determined on a dry-weight basis; on a wet-weight basis (as used in other studies) the high value of 20,000X would decrease to a BCF of 3100X.

Additional estimated and empirical BCF values were reported in USEPA, 2010. For green algae, empirical BCFs were a range of 2800X – 4400X while estimated (KABAM) BCFs were 4801X. For bluegill sunfish, empirical BCFs were a range of 960X – 1100X while the estimated (KABAM) BCF was 4806X. For golden orfe, empirical BCFs were a range of 1130X – 1140X while the estimated (KABAM) BCF was 4806X. For topmouth gudgeon, the empirical BCF was 238X while the estimated (KABAM) BCF was 4806X. For rainbow trout, empirical BCFs were a range of 90X – 950X while the estimated (KABAM) BCFs were 1646 – 8006X.

2.4.4. Environmental Transport Mechanisms

Potential transport mechanisms include pesticide surface water runoff, spray drift, and secondary drift of volatilized or soil-bound residues (dust) leading to deposition onto nearby or more distant ecosystems. Surface water runoff is expected to be the major route of exposure for most uses of PCNB; volatilization and spray drift is expected to be another major route for the foliar spray on turf use which does not include incorporation. Because PCNB and its major degradates are all generally persistent under field conditions, over time the compounds may be present in field runoff and could thus reach surface water bodies.

Spray drift levels will be affected by application methods. Except for turf, all applications are soil applied and incorporated at the time of planting. Greater spray drift will be associated with foliar applications (turf), while incorporated granular applications should result in relatively lower levels of drift. Applications of flowable product with immediate soil coverage or disking in may also result in slightly lower drift relative to foliar use.

PCNB is a moderately volatile compound that can be expected to dissipate through volatilization; however, this route of dissipation from the soil should be minimized for uses which include incorporation of the pesticide into the soil at application. Volatilization should be a more significant route of dissipation of the parent and the degradate PCA when PCNB is not incorporated, such as the foliar use on turf. Submitted data from a laboratory volatility study (MRID 41178001) showed that maximum volatilization occurred within 24 hours of treatment. Submitted data from a field volatility study (MRID 43751101) showed that volatilized PCNB and PCA were both at a maximum on day 0 and declined throughout the rest of the 13-day study (although large amounts of rain which fell during study days 3 to 9 may have altered the volatilization rate relative to that which might occur under more typical field conditions). It has been observed that PCNB may volatilize more from moist or saturated soils relative to dry ones due to decreased adsorption in the wetter soils (Casely, 1968). Casely (1968) reported that greater than 2/3 of the approximately 80% of the applied PCNB that was lost from soil during 10 months of incubation was due to volatilization, with the remaining losses attributed to microbial degradation. Data on volatilization from foliar surfaces were not available. Large Henry's Law constants for the degradates PCA, PCTA, PCB and PCP (on the order of 10^{-7} to 10^{-4} atm-m^3-mol) indicate that volatilization may also be an important environmental fate process for these compounds. Based on its vapor pressure (1.13×10^{-4} Torr at 25^{o}C), PCNB is expected to exist primarily in the vapor phase while in the atmosphere.

A number of studies have documented atmospheric transport and re-deposition of pesticides from the Central Valley to the Sierra Nevada Mountains (Fellers *et al.,* 2004; LeNoir *et al.,* 1999; McConnell *et al.,* 1998; Sparling *et al.,* 2001). PCNB was not monitored in these studies specifically, but is used in the area or may enter the area through atmospheric transport. Prevailing winds blow across the Central Valley eastward to the Sierra Nevada Mountains, transporting airborne industrial and agricultural pollutants into the Sierra Nevada ecosystems (Fellers *et al.,* 2004; LeNoir *et al.,* 1999; McConnell *et al.,* 1998). Several sections of the range and critical habitat for the BCB, CTS-CC, CTS-SC and CTS-SB, DS, CCR, CFWS, SFGS, and TG are located east of the Central Valley. The magnitude of transport via secondary drift depends on PCNB's ability to be mobilized into air and its eventual removal through wet and dry deposition of gases/particles and photochemical reactions in the atmosphere. Therefore, physicochemical properties of PCNB that describe its potential to enter the air from water or soil (*e.g.,* Henry's Law constant and vapor pressure), pesticide use data, modeled estimated concentrations in water and air, and available air monitoring data from the Central Valley and the Sierra Nevadas are important for evaluating the potential for atmospheric transport of PCNB to locations where it could impact the species listed above.

In general, deposition of drifting or volatilized pesticides is expected to be greatest close to the site of application. The potential impact away from the PCNB use sites is affected by the environmental concentrations, which are generally expected to decrease with distance. While computer models may be useful for some pesticides in determining a distance beyond which the EECs will fall below an LOC for a taxonomic group, model limitations preclude a quantitative estimate of exposure for PCNB.

Additionally, PCNB can undergo long-range transport (see Section 3.2.4.d). The long-range transport of PCNB has been previously assessed by the Agency using the OECD Pov and LRTP Screening Tool (v2.0)[1].
Results from the OECD Tool do not indicate absolute loading of pesticides in the environment but help to compare the inherent characteristics with reference pesticides identified by the Stockholm Convention as Persistent Organic Pollutants (POPs) according to their overall persistence, characteristic travel distance (CTD) and transfer efficiency (TE). In this case, Aldrin and HCB are used as reference chemicals.

CTD represents the potential of a chemical to be transported over long distances in air or water. In the OECD Tool, CTD is the distance at which the concentration of a chemical decreases to 37% due to transport of chemical by a constant flow of air (wind speed of 0.02 m/s) or water (ocean water circulation speed of 0.02m/s (Scheringer *et al.,* 2006). Based on modeling results obtained using the screening tool, it was determined that PCNB has comparable or higher long range transport potential compared to chemicals that are known to move via long range transport. Characteristic travel distances for PCNB, PCA, PCTA and PCB were calculated to be 24,400 km (15,161 miles), 3900 km (2423 miles), 2131 km (1324 miles), and 43,400 km (26,968miles), respectively.

A more complete discussion of the OECD screening tool and the determination of the potential for long-range transport of PCNB and its major degradates may be found in the USEPA, 2010 document " *2008 Science Advisory Panel Meeting Follow Up: Assessment of the Bioaccumulation and Long-Range Transport Potential (LRTP) and of Pentachloronitrobenzene (PCNB) and Associated Ecological Risks*."

2.4.5. Mechanism of Action

The chemical's mode of action is the inhibition of hyphal growth by means of the competitive inhibition of inositol. PCNB is used to control (preventative and residual) soil-borne diseases prior to infection of the plants. To date, resistance to PCNB has reportedly not been observed.

2.4.6. Use Characterization

Analysis of labeled use information is the critical first step in evaluating the federal action. The current labels for PCNB represent the FIFRA regulatory action; therefore, labeled use and application rates specified on the label form the basis of this assessment. The assessment of use information is critical to the development of the action area and selection of appropriate modeling scenarios and inputs.

Although the RED (USEPA, 2006) found several uses of PCNB ineligible for reregistration, only a subset of the RED mitigation measures have been implemented. The uses of PCNB on golf

[1] Available at http://www.oecd.org/document/17/0,3343,en_2649_34373_40754961_1_1_1_1,00.html

course roughs, residential sites including lawns, yards, and ornamental plants and gardens around homes and apartments, grounds around day care facilities; school yards, parks (except industrial parks), playgrounds, and athletic fields (except professional and collegiate athletic fields) were terminated via the Section 6(f) process of FIFRA, effective July 15, 2009. All legal sale, distribution, and formulation (from manufacturing-use products) of existing stocks were to have ceased by January 2011, but the sale and distribution of these existing stocks was stopped in August 2010, when the EPA Office of Enforcement and Compliance Assurance (OECA) issued a Stop Sale, Use, or Removal Order (SSURO) for the technical registrant's PCNB products based on the failure of the technical registrant to document the presence of impurities of toxicological concern in its technical grade PCNB. The SSURO is no longer in effect at the time of this assessment, having been vacated by court ruling on August 17, 2011.

Several subsequent actions outside of reregistration (related to risks associated with impurities in the PCNB technical) have resulted in application rate reductions, prohibitions against certain methods of application, and use terminations that have been implemented or requested by the registrant under Section 6(f) of FIFRA. All these actions are described in the "Verification Memo for SF Bay Species" (Appendix A) and the associated "Use Closure Memo" further detailing current uses (Appendix B).

Although seed treatments were determined to be eligible for reregistration, the registrant has chosen not to support the seed treatment uses. Other registered uses which are not currently being marketed and/or the uses have not been deleted from this label through a 6(f) process are as follows: beans, garlic, peanuts, peppers, tomatoes, seed treatment (barley, beans, corn, cotton, oats, peanuts peas, rice safflower, sorghum, soybeans, sugar beets, and wheat), ornamentals (African violets, azaleas, bedding plants, begonias, calendula, camellia, carnation chrysanthemum, gladiolus, hyacinth, iris, larkspur, lilies, narcissus, ornamental flowering plants, woody ornamentals, southern pine seed or seedlings, poinsettias, roses, snapdragon, southern magnolia trees, sweet peas, tulips), turf on sod farms, industrial parks, and professional and college athletic fields. Although these uses are currently registered, the registrant is restricted from marketing these uses at this time. Therefore, these uses are not assessed in this document.

On November 23, 2011, the Agency approved registration amendments requested by the technical registrant that resulted in the following amendment to the terms and conditions of the registration: 1) Amvac agreed not to sell, distribute, or formulate any Technical Grade PCNB (EPA Reg. # 5481-197) with concentrations of dioxin greater than a particular limit; 2) The technical registrant agreed to market only three PCNB products (labeled to conform to the other elements of the approved amendments)--technical grade PCNB, Turfcide® 10% Granular (Reg. # 5481-8988) and Turfcide® 4F (Reg. # 5481-8992). 3) The technical registrant agreed to request cancellation of all non-liquid/non-granular registrations of PCNB. Any changes in application rate or labeled uses since the 2006 RED are not reflected in the BEAD's Label Use Information System (LUIS) EFED Label Data Report.

The Agency published, on May 2, 2012, a cancellation order for 15 PCNB products for which the technical registrant, Amvac, previously requested cancellation. The cancellation was effective on the date of publication. The affected products are the non-liquid/non-granular end-use formulations that Amvac does not intend to support. The order will prohibit the sale, distribution, and formulation of existing stocks of the affected products effective upon publication of the order.

The only currently approved uses for which the granular formulation is labeled are: golf course turf (TGF); broccoli, Brussels sprouts, cabbage, cauliflower, collards, kale, and mustard greens (GA only); cotton; and potato (not in CA). The only currently approved uses for which the flowable formulation is labeled are: golf course turf (GTF), broccoli, Brussels sprouts, cabbage, cauliflower, cotton, potato, and ornamental bulbs (dip).

Aerial, chemigation, and hand-held application are prohibited. Turfcide® 4F may be applied by bulb soak or groundboom application only, and Turfcide® 10% Granular may be applied by tractor-drawn spreaders only. Except for turf and bulbs, all applications are soil applied and incorporated at the time of planting. All these are in-furrow applications, except for cole crops, which may be band-row applied or broadcast and then disked in. Bulbs are soaked in a solution of PCNB prior to planting. All applications to golf course turf must be followed by ¼" of irrigation water. Applications to ornamental bulbs are currently limited to bulb soak applications only.

In December 12, 2011, the PCNB technical registrant requested the amendment of its PCNB registrations to allow use on: containerized stock of broccoli, Chinese broccoli, Brussels sprouts, cabbage, Chinese cabbage, cauliflower, collards, kale, and mustard; commercial/industrial/ residential lawns; sod farms; peanut; beans; garlic; magnolia tree (foliar spray); tomato; pepper; southern pine (seed orchard); and ornamentals (shade trees, herbaceous, woody shrubs, and vines). The due date for the Agency's decision is August12, 2012.

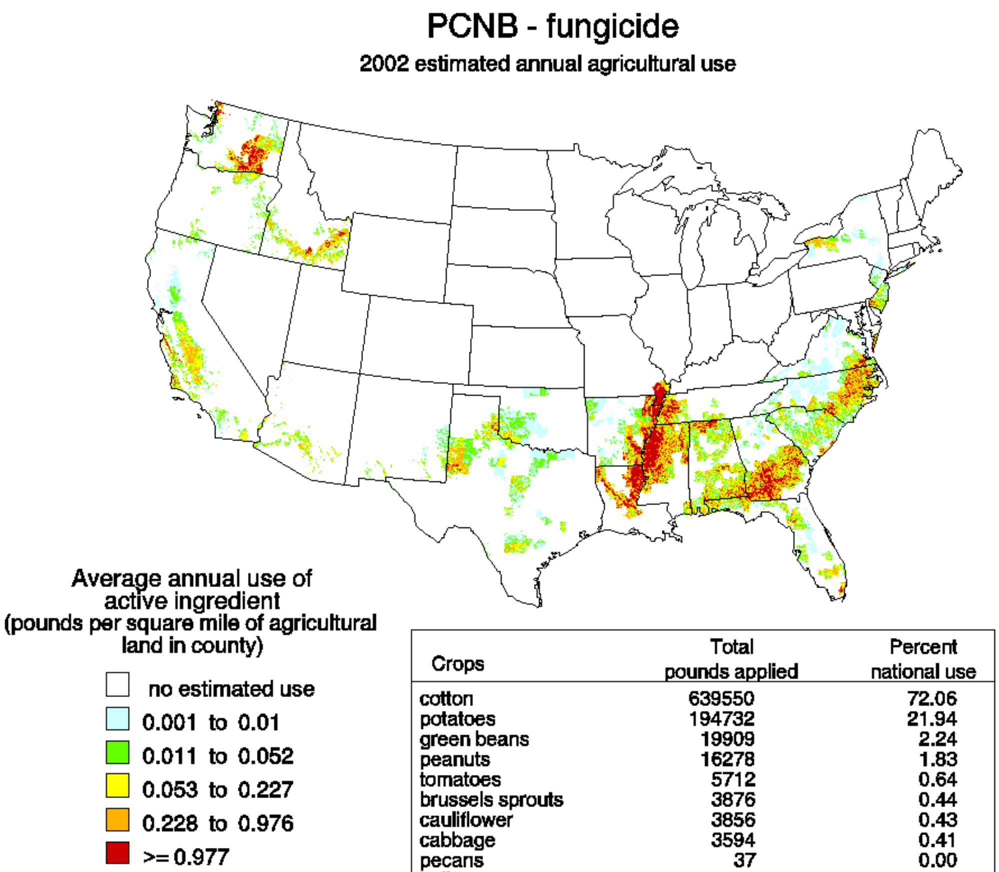
Table 2-3 presents the currently approved uses and corresponding application rates and methods of application considered in this assessment. Additional labeled uses for collards, kale, and mustard greens are not assessed, as they are restricted to GA. Also, there is a higher label rate for the potato use, but it is not allowed in California.

Table 2-3. PCNB Uses Assessed for California

Use (Application Method)	Form.	Maximum Single App. Rate (lbs a.i./acre)	Maximum App. Rate per Year (lbs a.i./acre)	Maximum Number of App. per Year	Minimum Retreatment Interval (days)
Cole crops[1](ground spray; banded)	Flowable & granular	22.5	22.5	1 (at planting)	NA
Cotton (ground spray; in-furrow and on surface)	Flowable & granular	2.0	2.0	1 (at planting)	NA
Potatoes	Flowable	5	5	1 (at planting)	NA
Turf (foliar spray)	Flowable	33	66	2	28
Turf (ground spreader)	Granular	43.56	87.12	2	28
Ornamental bulb dip	Flowable	Not specified on label.	Not specified on label.	(at planting)	NA

Abbreviations: App. = applications; Form. = formulation; NA = Not applicable. [1]Represents all cole crops, but information presented is based on one crop of Brussels Sprouts grown annually.

According to the United States Geological Survey's (USGS) national pesticide usage data (based on information from 1999 to 2004), an average of 887,548 lbs of PCNB is applied nationally to agricultural use sites in the U.S. (non-agricultural uses are not included) (Figure 2-2). During those years, cotton and potatoes accounted for 94% of agricultural crop usage, with most of the remaining agricultural crop use on green beans, peanuts, tomatoes, and cole crops. Based on information presented in the PCNB RED (USEPA, 2006), turf and seed treatment were also predominant uses. Of these previously identified main uses of PCNB, the only currently approved uses are cotton, potatoes, cole crops and turf (golf course GTF).



Crops	Total pounds applied	Percent national use
cotton	639550	72.06
potatoes	194732	21.94
green beans	19909	2.24
peanuts	16278	1.83
tomatoes	5712	0.64
brussels sprouts	3876	0.44
cauliflower	3856	0.43
cabbage	3594	0.41
pecans	37	0.00
bell peppers	4	0.00

Figure 2-2. PCNB Use in Total Pounds per County (from http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=02&map=m5021)**[2]**

[2] The pesticide use maps available from this site show the average annual pesticide use intensity expressed as average weight (in pounds) of a pesticide applied to each square mile of agricultural land in a county. The area of each map is based on state-level estimates of pesticide use rates for individual crops that were compiled by the CropLife Foundation, Crop Protection Research Institute based on information collected during 1999 through 2004 and on 2002 Census of Agriculture county crop acreage. The maps do not represent a specific year, but rather show typical use patterns over the five year period 1999 through 2004.

California State law requires that every agricultural pesticide application be reported to the state and made available to the public. Under California law, agricultural uses reported include pesticide applications to parks, golf courses, cemeteries, rangeland, pastures, and along roadside and railroad rights-of-way. According the California Department of Pesticide Regulation, the primary uses not required to be reported include "home-and-garden uses and most industrial and institutional uses" (http://www.cdpr.ca.gov/docs/pur/purmain.htm). Therefore, CDPR PUR is considered the most comprehensive source of pesticide usage data for the state and includes both agricultural and non-agricultural sites.

The Agency's Biological and Economic Analysis Division (BEAD) provides an analysis of both national- and county-level usage information (USEPA, 2012) using state-level usage data obtained from USDA-NASS[3], Doane (www.doane.com; the full dataset is not provided due to its proprietary nature) and the California's Department of Pesticide Regulation Pesticide Use Reporting (CDPR PUR) database[4]. CDPR PUR is considered a more comprehensive source of usage data than USDA-NASS or EPA proprietary databases, and thus the usage data reported for PCNB by county in this California-specific assessment were generated using CDPR PUR data. Twelve years (1999-2010) of usage data were included in this analysis. Data from CDPR PUR were obtained for every agricultural pesticide application made on every use site at the section level (approximately one square mile) of the public land survey system. BEAD summarized these data to the county level by site, pesticide, and unit treated. Calculating county-level usage involved summarizing across all applications made within a section and then across all sections within a county for each use site and for each pesticide. The county level usage data that were calculated include: average annual pounds applied, average annual area treated, and average and maximum application rate across all twelve years. The units of area treated are also provided where available. Between 1999 and 2010, annual use of PCNB in California ranged from approximately 24,018 to 67,423 pounds a.i. Generally, there was a decreasing trend in use over this time period, although in 2010 the annual use (37,018 lbs) increased back to levels seen in 2003-2005. The CA counties with the largest usage of PCNB in terms of total pounds applied from 1999-2010 are Contra Costa, Del Norte, Fresno, Kern, Kings, Los Angeles, Merced, Monterey, Orange, Riverside, San Diego, San Luis Obispo, San Mateo, Santa Barbara, Santa Clara, Santa Cruz, and Sonoma. Each of these counties had total applications (for the time period 1999-2010) ranging from approximately 10,000 to 30,000 pounds, with the exception of San Mateo, which had a usage total of approximately 79,000 pounds.

A summary of PCNB usage information for all California use sites is provided below in **Table 2-4**, while **Table 2-5** provides a summary of CDPR use information for the years 1999-2010 for all currently labeled uses of PCNB (except bulbs) in California.

Based on information provided by BEAD, there is limited (if any) commercial production of bulbs in the San Francisco Bay watershed.

---

[3] United States Depart of Agriculture (USDA), National Agricultural Statistics Service (NASS) Chemical Use Reports provide summary pesticide usage statistics for select agricultural use sites by chemical, crop and state. See http://www.pestmanagement.info/nass/app_usage.cfm.

[4] The California Department of Pesticide Regulation's Pesticide Use Reporting database provides a census of pesticide applications in the state. See (http://www.cdpr.ca.gov/docs/pur/purmain.htm).

Table 2-4. Summary of California Department of Pesticide Registration (CDPR) Pesticide Use Reporting (PUR) Data for Average and Maximum Application Rates from 1999 to 2010 for Currently Registered PCNB Uses[1]

Site Name	Average Application Rate (lbs a.i./A)	Maximum Application Rate (lbs a.i./A)
Broccoli	2.1	15.6
Brussels Sprout	21.6	90.0
Cabbage	3.7	15.6
Cauliflower	3.7	4.5
Cotton	1.6	1.9
Cotton	0.1	0.1
Cotton	0.5	1.7
Potato	2.5	4.0
Turf/Sod	14.8	75.8

[1] Based on data supplied by BEAD (USEPA, 2012) based on data obtained from (http://www.cdpr.ca.gov/docs/pur/purmain.htm).

Table 2-5. **Summary of California Department of Pesticide Registration (CDPR) Pesticide Use Reporting (PUR) Data for Total Pounds PCNB Applied from 1999 to 2010 for Currently Registered PCNB Uses[1]**

Site Name	Total Pounds Applied											
	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Broccoli	0	0	0	1.5	0	18.8	0	0	0	0	0	0
Broccoli	628	2	0	6	13	7	38	1	2	0	153	0
Brussels Sprout	8,843	2,970	4,819	5,435	6,416	5,614	4,854	3,862	5,760	8,187	4,390	5,751
Cabbage	1,056	0	0	0	0	0	0	0	0	0	0	0
Cauliflower	5,114	4,707	240	0	0	0	92	0	0	0	0	0
Cotton	467	244	192	1,022	1,665	1,141	2,724	2,009	1,013	0	0	0
Cotton	0	0	15.29	5.63	0	0	0	0	0	0	0	0
Cotton	416	76	0.7	0	0	0	0	0	0	0	0	0
Cotton	13,241	15,308	11,743	4,145	1,975	966	902	45	77	0	0	0
Potato	0	0	0	704.49	725	1,058	0	0	0	372	0	0
Turf/Sod	0	0	0	0	0	0	40	0	0	0	0	11
Turf/Sod	60	50	0	0	15	0	0	0	0	56	0	4
Turf/Sod	111	463	25	298	30	178	616	26	72	0	14	70

[1] Based on data supplied by BEAD (USEPA, 2012) based on data obtained from (http://www.cdpr.ca.gov/docs/pur/purmain.htm). All values were rounded to whole numbers.

2.5. Assessed Species

Table 2-6 provides a summary of the current distribution, habitat requirements, and life history parameters for the listed species being assessed. More detailed life-history and distribution information can be found in **Attachment II**. See **Figure 2-3** for maps of the current range and designated critical habitat, if applicable, of the assessed listed species. See Section 2.1 for information on when each species was listed and a general description of their ranges.

Table 2-6. Summary of Current Distribution, Habitat Requirements, and Life History Information for the Assessed Listed Species[1]

Assessed Species	Size	Current Range	Habitat Type	Designated Critical Habitat?	Reproductive Cycle	Diet
San Francisco Garter Snake (SFGS) (*Thamnophis sirtalis tetrataenia*)	Adult (46-131 cm in length), Females – 227 g, Males – 113 g; Juveniles – 2 g (Cover Jr. and Boyer, 1988) (18–20 cm in length)	San Mateo County	Densely vegetated freshwater ponds near open grassy hillsides; emergent vegetation; rodent burrows	No	Oviparous Reproduction[2] Breeding: Spring (Mar. and Apr.) and Fall (Sept. to Nov.) Ovulation and Pregnancy: Late spring and early summer Young: Born 3-4 months after mating	Juveniles: frogs (Pacific tree frog, CRLF, and bullfrogs depending on size) and insects Adults: primarily frogs (mainly CRLFs; also bullfrogs, toads); to a lesser extent newts; freshwater fish and invertebrates; insects and small mammals
California Clapper Rail (CCR) (*Rallus longirostris obsoletus)*	250 - 350 g Juveniles ~50 g[3]	Alameda, Contra Costa, Marin, Napa, San Francisco, San Mateo, Santa Clara, Solano, and Sonoma counties	Tidal marsh habitat	No	Breeding: Feb. - August Nesting: mid-March-Aug. Lay Eggs: March - July Incubation: 23 to 29 days; Leave nest: 35 to 42 days after hatch; Juveniles fledge at ten weeks and can breed during the spring after they hatch	Opportunistic feeders: freshwater and estuarine invertebrates, seeds, worms, mussels, snails, clams, crabs, insects, and spiders; occasionally consume small birds and mammals, dead fish, up to 15% plant material
Bay Checkerspot Butterfly (BCB) (*Euphydryas editha bayensis*)	Adult butterfly - 5 cm in length	Santa Clara and San Mateo Counties [Because the BCB distribution is considered a metapopulation, any site with appropriate habitat in the vicinity of its historic range (Alameda, Contra Costa, San Francisco, San Mateo, and Santa Clara counties) should be considered potentially occupied by the butterfly (USFWS	1) Primary habitat – native grasslands on large serpentine outcrops; 2) Secondary habitat – ‘islands’ of smaller serpentine outcrops with native grassland; 3) Tertiary habitat – non-serpentine areas	Yes	Larvae hatch in March – May and grow to the 4th instar in about two weeks. The larvae enter into a period of dormancy (diapause) that lasts through the summer. The larvae resume activity with the start of the rainy season. Larvae pupate	Obligate with dwarf plantain. Primary diet is dwarf plantain plants (may also feed on purple owl’s-clover or exserted paintbrush if the dwarf plantains senesce before the larvae pupate). Adults feed on the nectar of a

Assessed Species	Size	Current Range	Habitat Type	Designated Critical Habitat?	Reproductive Cycle	Diet
		1998, p. II-177)].	where larval food plants occur		once they reach a weight of 300 - 500 milligrams. Adults emerge within 15 to 30 days depending on thermal conditions, feed on nectar, mate and lay eggs during a flight season that lasts 4 to 6 weeks from late February to early May	variety of plants found in association with serpentine grasslands
California Tiger Salamander (CTS) (*Ambystoma californiense*)	Adult 14.2-80.5 g[4]	CTS-SC are primarily found on the Santa Rosa Plain in Sonoma County. CTS-CC occupies the Bay Area (central and southern Alameda, Santa Clara, western Stanislaus, western Merced, and the majority of San Benito Counties), Central Valley (Yolo, Sacramento, Solano, eastern Contra Costa, northeast Alameda, San Joaquin, Stanislaus, Merced, and northwestern Madera Counties), southern San Joaquin Valley (portions of Madera, central Fresno, and northern Tulare and Kings Counties), and the Central Coast Range (southern Santa Cruz, Monterey, northern San Luis Obispo, and portions of western San Benito, Fresno, and Kern Counties). CTS-SB are found in Santa Barbara County.	Freshwater pools or ponds (natural or man-made, vernal pools, ranch stock ponds, other fishless ponds); Grassland or oak savannah communities, in low foothill regions; Small mammal burrows	Yes	Emerge from burrows and breed: fall and winter rains Eggs: laid in pond Dec. – Feb., hatch: after 10 to 14 days Larval stage: 3-6 months, until the ponds dry out, metamorphose late spring or early summer, migrate to small mammal burrows	Aquatic Phase: algae, snails, zooplankton, small crustaceans, and aquatic larvae and invertebrates, smaller tadpoles of Pacific tree frogs, CRLF, toads; Terrestrial Phase: terrestrial invertebrates, insects, frogs, and worms
Tidewater Goby (TG)	50 mm in length	Along the coast in California (from 3 miles south of the CA/OR border	Coastal brackish water habitats,	Yes	They are typically an annual species. Spawning	They are generalists that eat a wide variety

Assessed Species	Size	Current Range	Habitat Type	Designated Critical Habitat?	Reproductive Cycle	Diet
(*Eucyclogobius newberryi*)		to 44 miles north of the US/Mexico border –there are gaps in the geographic distribution where lagoons and/or estuaries are absent)	primarily coastal lagoons, estuaries, river mouths, and marshes. They are typically found in water less than 1 m deep with salinities of less than 12 parts per thousand.		has been observed in every month of the year except Dec. Females may lay more than 1 clutch in a year. Eggs take from 9 to 11 days to hatch.	of invertebrates [small benthic invertebrates, crustaceans, snails, mysids, and aquatic insect larvae). Juveniles probably feed on unicellular phytoplankton or zooplankton.
Delta Smelt (DS) (*Hypomesus transpacificus*)	Up to 120 mm in length	Suisun Bay and the Sacramento-San Joaquin estuary (known as the Delta) near San Francisco Bay, CA	The species is adapted to living in fresh and brackish water. They typically occupy estuarine areas with salinities below 2 parts per thousand (although they have been found in areas up to 18ppt). They live along the freshwater edge of the mixing zone (saltwater-freshwater interface).	Yes	They spawn in fresh or slightly brackish water upstream of the mixing zone. Spawning season usually takes place from late March through mid-May, although it may occur from late winter (Dec.) to early summer (July-August). Eggs hatch in 9 – 14 days.	They primarily planktonic copepods, cladocerans, amphipods, and insect larvae. Larvae feed on phytoplankton; juveniles feed on zooplankton.
California Freshwater Shrimp (CFWS) (*Syncaris pacifica*)	Up to 50 mm postorbital length (from the eye orbit to tip of tail)	Marin, Napa, and Sonoma Counties, CA	Freshwater, perennial streams; they prefer quiet portions of tree-lined streams with underwater vegetation and exposed tree roots	No	Breed once a year, typically in Sept. Eggs adhere to the pleopods and are cared for for 8 – 9 months; embryos emerge during May or early June.	Feed on detritus (algae, aquatic macrophyte fragments, zooplankton, and periphyton)

[1] For more detailed information on the distribution, habitat requirements, and life history information of the assessed listed species, see Attachment II.
[2] Oviparous = eggs hatch within the female's body and young are born live.
[3] No data on juvenile CCR body weights are available at this time. As a surrogate for CCR juveniles, data on captive 21-day king rails were averaged for the juvenile body weight. King rails make an appropriate proxy for the CCR in the absence of information. The birds were once considered the same species by taxonomists, are members of the same genus (*Rallus*), and occasionally interbreed where habitats overlap.
[4] See Page 369 of Trenham *et al.,* (2000).

Figure 2-3. Bay Checkerspot Butterfly Critical Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.

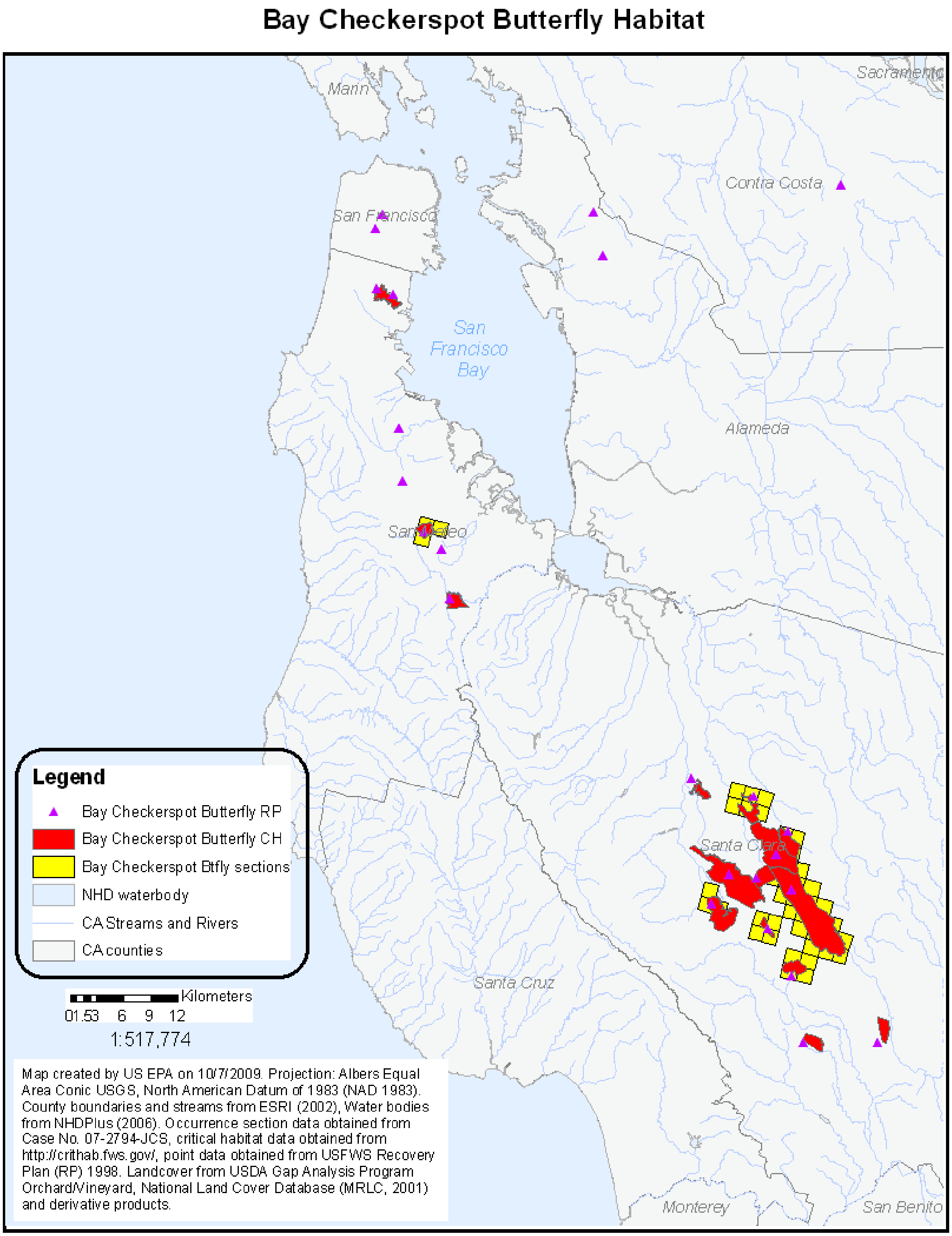


Figure 2-4. California Clapper Rail Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.

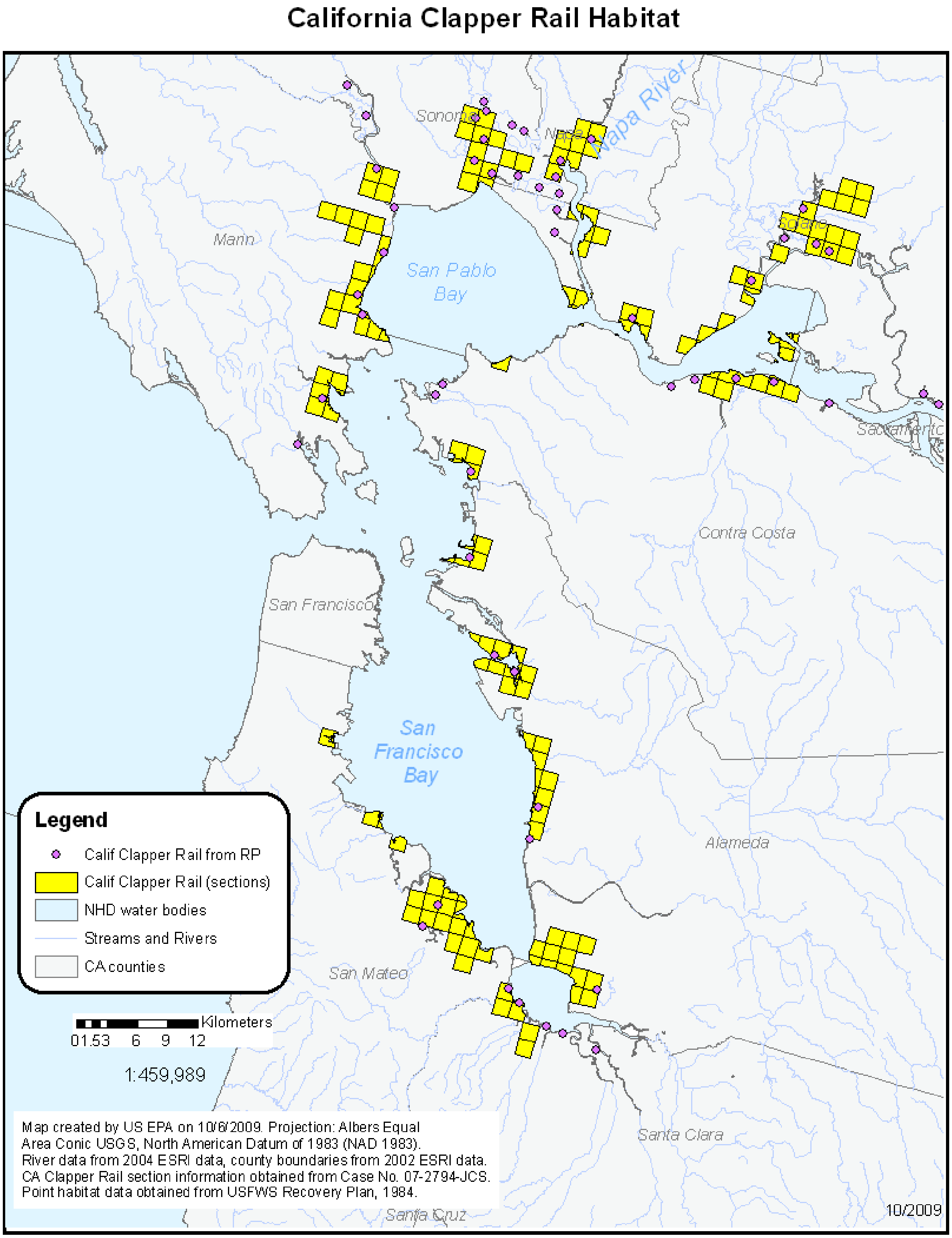


Figure 2-5. California Freshwater Shrimp Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.

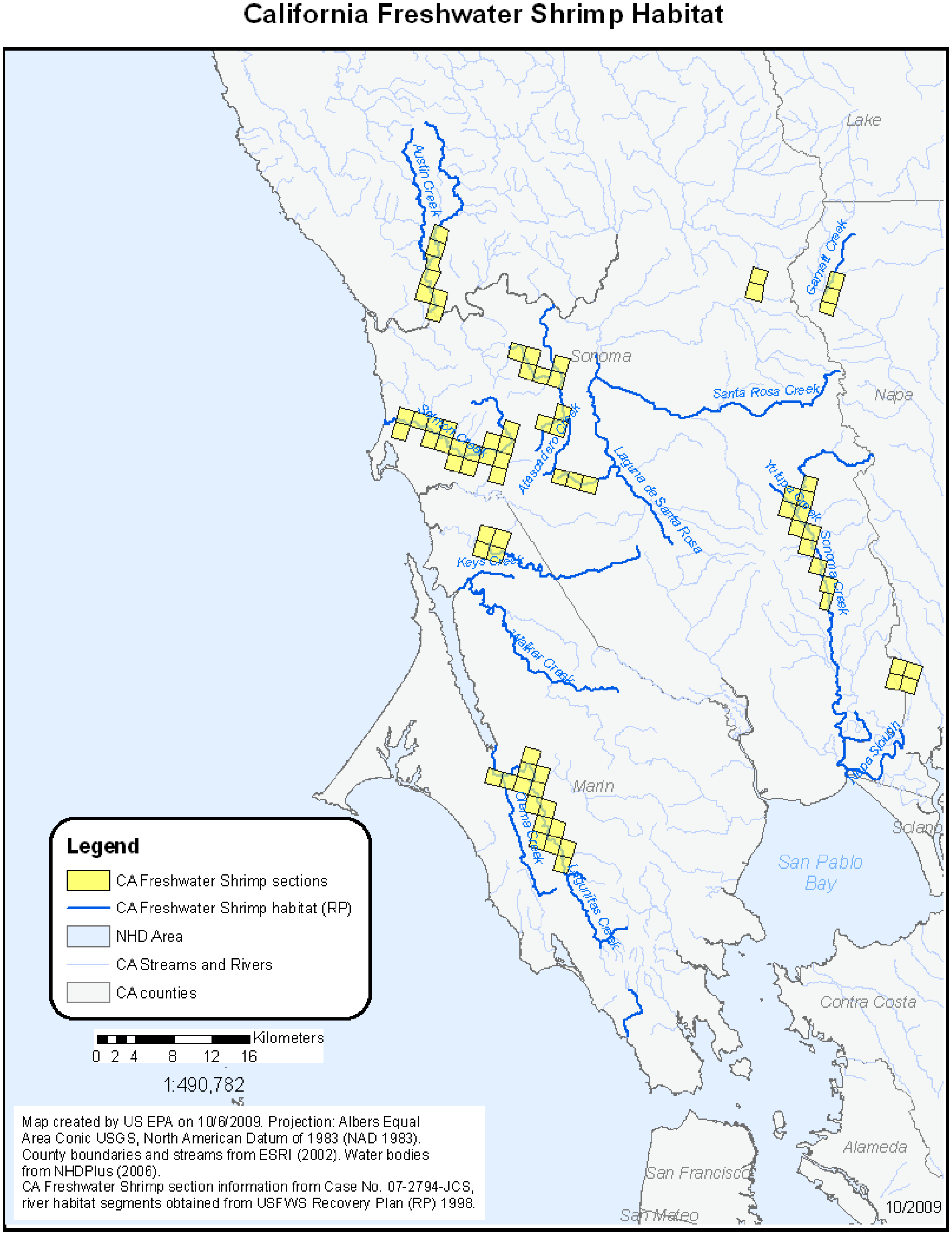


Figure 2-6. California Tiger Salamander Critical Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.

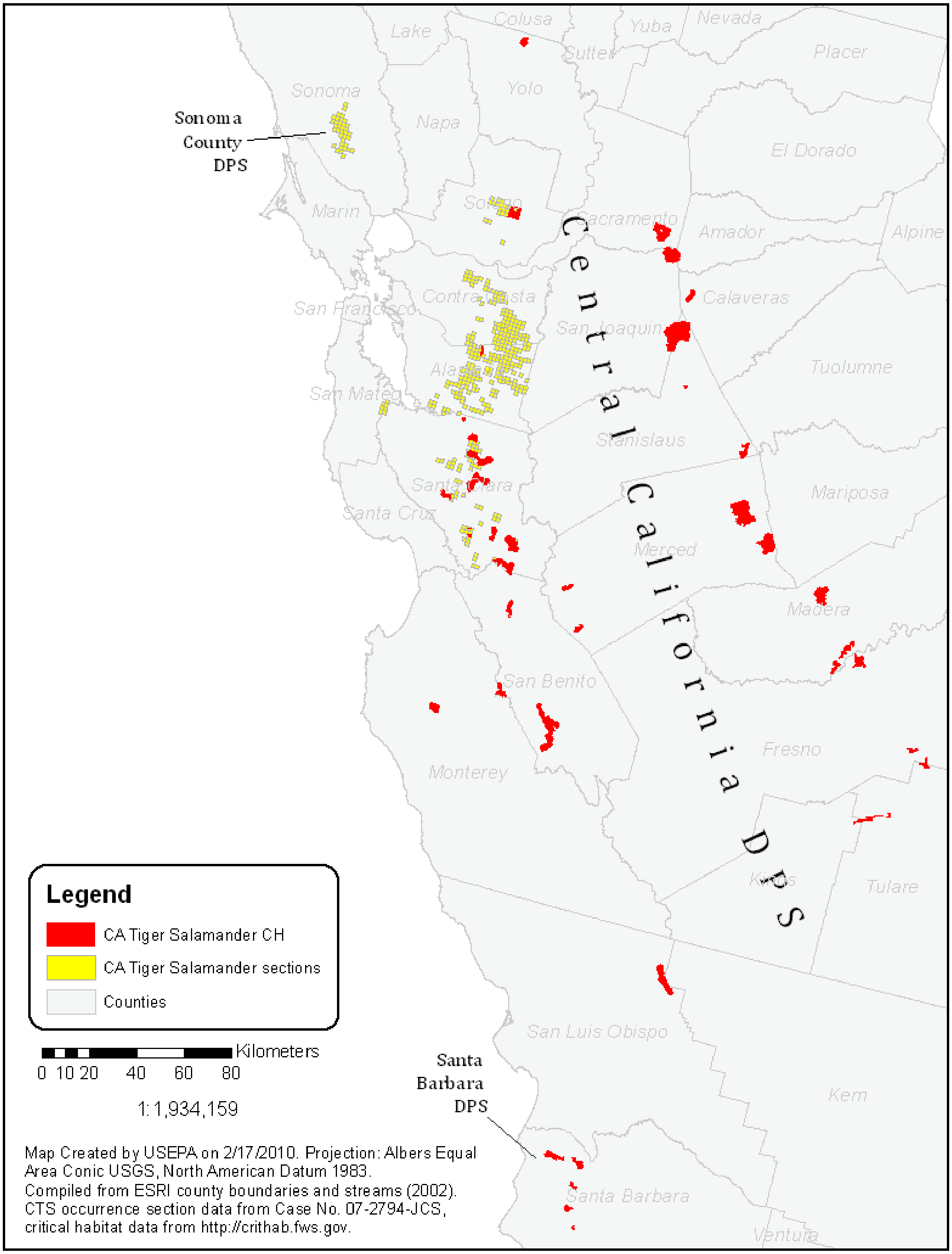


Figure 2-7. San Francisco Garter Snake Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.

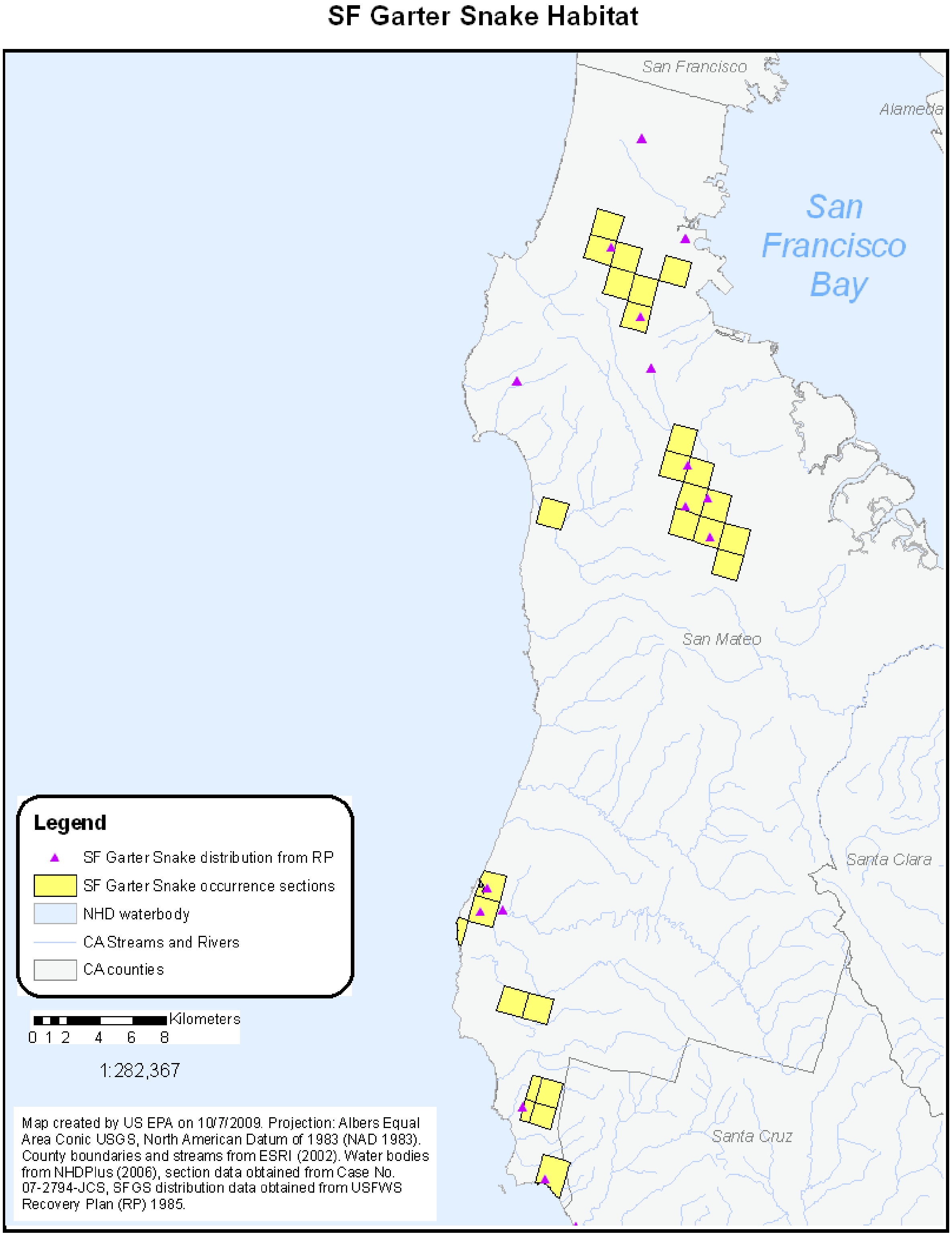
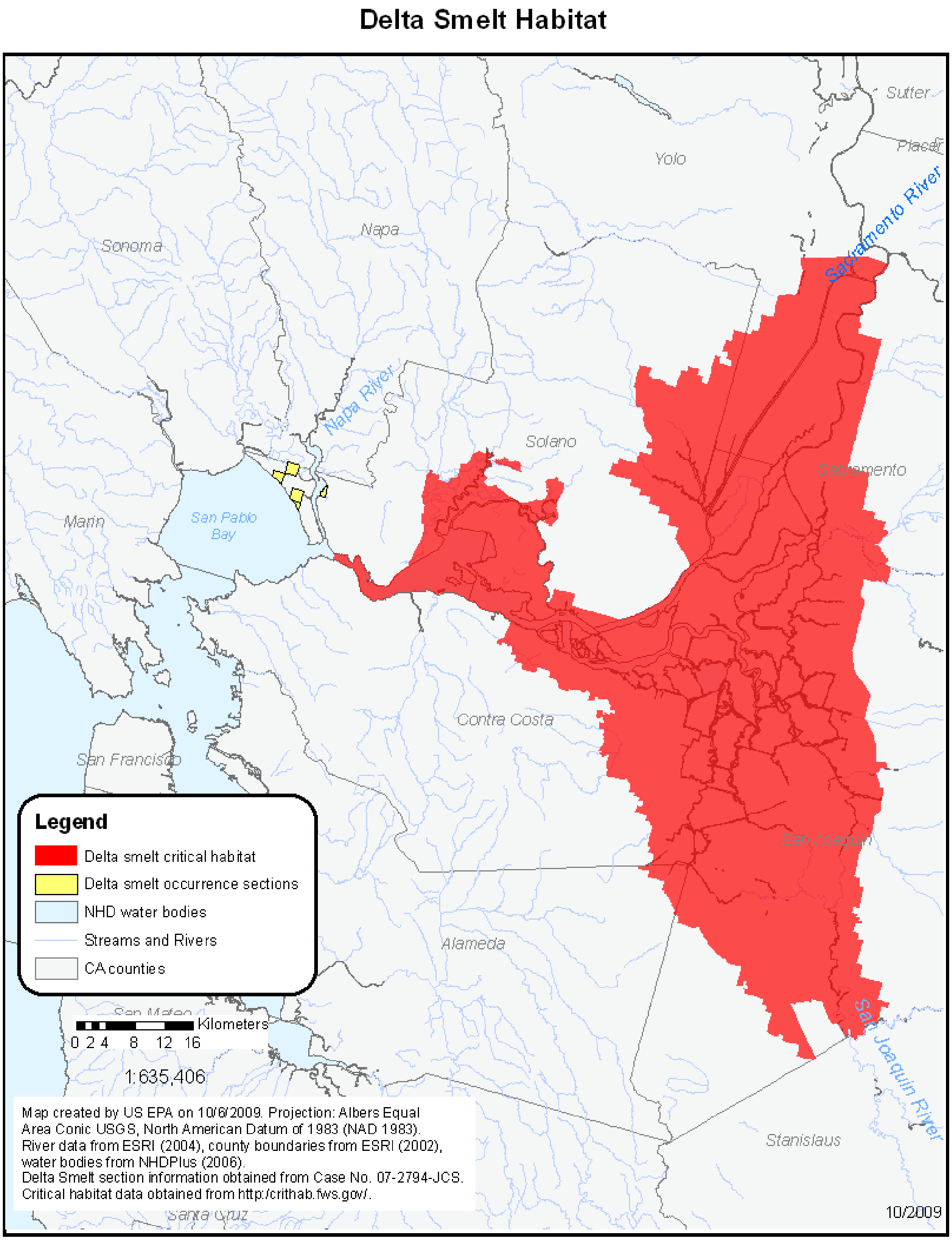


Figure 2-8. Delta Smelt Habitat and Occurrence Sections identified in Case No. 07-2794-JCS.



2.6. Designated Critical Habitat

Critical habitat has been designated for the BCB, CTS-CC, CTS-SB, DS, and TG. Risk to critical habitat is evaluated separately from risk to effects on the species. 'Critical habitat' is defined in the ESA as the geographic area occupied by the species at the time of the listing where the physical and biological features necessary for the conservation of the species exist, and there is a need for special management to protect the listed species. It may also include areas outside the occupied area at the time of listing if such areas are 'essential to the conservation of the species. Critical habitat designations identify, to the extent known using the best scientific and commercial data available, habitat areas that provide essential life cycle needs of the species or areas that contain certain primary constituent elements (PCEs) (as defined in 50 CFR 414.12(b)). **Table 2-7** describes the PCEs for the critical habitats designated for the BCB, CTS-CC, CTS-SB, DS, and TG.

Table 2-7. Designated Critical Habitat PCEs for the Bay Checkerspot Butterfly, California Tiger Salamander, Central California and Santa Barbara Distinct Population Segments, Delta Smelt, and Tidewater Goby[1]

Species	PCEs	Reference
California tiger salamander	Standing bodies of fresh water, including natural and man-made (*e.g.*, stock) ponds, vernal pools, and dune ponds, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a sufficient length of time (*i.e.*, 12 weeks) necessary for the species to complete the aquatic (egg and larval) portion of its life cycle[2]	FR Vol. 69 No. 226 CTS, 68584, 2004
	Barrier-free uplands adjacent to breeding ponds that contain small mammal burrows. Small mammals are essential in creating the underground habitat that juvenile and adult California tiger salamanders depend upon for food, shelter, and protection from the elements and predation	
	Upland areas between breeding locations (PCE 1) and areas with small mammal burrows (PCE 2) that allow for dispersal among such sites	
Bay Checkerspot Butterfly	The presence of annual or perennial grasslands with little to no overstory that provide north/south and east/west slopes with a tilt of more than 7 degrees for larval host plant survival during periods of atypical weather (*e.g.*, drought).	66 FR 21449 21489, 2001
	The presence of the primary larval host plant, dwarf plantain (*Plantago erecta*) (a dicot) and at least one of the secondary host plants, purple owl's-clover or exserted paintbrush, are required for reproduction, feeding, and larval development.	
	The presence of adult nectar sources for feeding.	
	Aquatic features such as wetlands, springs, seeps, streams, lakes, and ponds and their associated banks, that provide moisture during periods of spring drought; these features can be ephemeral, seasonal, or permanent.	
	Soils derived from serpentinite ultramafic rock (Montara, Climara, Henneke, Hentine, and Obispo soil series) or similar soils (Inks, Candlestick, Los Gatos, Fagan, and Barnabe soil series) that provide areas with fewer aggressive, nonnative plant species for larval host plant and adult nectar plant survival and reproduction.[2]	
	The presence of stable holes and cracks in the soil, and surface rock outcrops that provide shelter for the larval stage of the bay	

Species	PCEs	Reference
	checkerspot butterfly during summer diapause.[2]	
Tidewater Goby	Persistent, shallow (in the range of about 0.1-2 m), still-to-slow-moving, aquatic habitat most commonly ranging in salinity from less than 0.5 ppt to about 10-12 ppt, which provides adequate space for normal behavior and individual and population growth	65 FR 69693 69717, 2000
	Substrates (*e.g.*, sand, silt, mud) suitable for the construction of burrows for reproduction	
	Submerged and emergent aquatic vegetation, such as *Potamogeton pectinatus* and *Ruppia maritima*, that provides protection from predators	
	Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity.	
Delta Smelt	Spawning Habitat—shallow, fresh or slightly brackish backwater sloughs and edgewaters to ensure egg hatching and larval viability. Spawning areas also must provide suitable water quality (*i.e.*, low "concentrations of pollutants) and substrates for egg attachment (*e.g.*, submerged tree roots and branches and emergent vegetation).	59 FR 65256 65279, 1994
	Larval and Juvenile Transport—Sacramento and San Joaquin Rivers and their tributary channels must be protected from physical disturbance and flow disruption. Adequate river flow is necessary to transport larvae from upstream spawning areas to rearing habitat in Suisun Bay. Suitable water quality must be provided so that maturation is not impaired by pollutant concentrations.	
	Rearing Habitat—Maintenance of the 2 ppt isohaline and suitable water quality (low concentrations of pollutants) within the Estuary is necessary to provide delta smelt larvae and juveniles a shallow protective, food-rich environment in which to mature to adulthood.	
	Adult Migration— Unrestricted access to suitable spawning habitat in a period that may extend from December to July. Adequate flow and suitable water quality may need to be maintained to attract migrating adults in the Sacramento and San Joaquin River channels and their associated tributaries. These areas also should be protected from physical disturbance and flow disruption during migratory periods.	

[1] These PCEs are in addition to more general requirements for habitat areas that provide essential life cycle needs of the species such as, space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, rearing (or development) of offspring; and habitats that are protected from disturbance or are representative of the historic geographical and ecological distributions of a species.
[2] PCEs that are abiotic, including, physical-chemical water quality parameters such as salinity, pH, and hardness are not evaluated.

More detail on the designated critical habitat applicable to this assessment can be found in Attachment II. Activities that may destroy or adversely modify critical habitat are those that alter the PCEs and jeopardize the continued existence of the species. Evaluation of actions related to use of PCNB that may alter the PCEs of the designated critical habitat for the BCB, CTS-CC, CTS-SB, DS, and TG form the basis of the critical habitat impact analysis.

As previously noted in Section 2.1, the Agency believes that the analysis of direct and indirect effects to listed species provides the basis for an analysis of potential effects on the designated critical habitat. Because PCNB is expected to directly impact living organisms within the action area, critical habitat analysis for PCNB is limited in a practical sense to those PCEs of critical habitat that are biological or that can be reasonably linked to biologically mediated processes.

2.7. Action Area and LAA Effects Determination Area

2.7.1. Action Area

The action area is used to identify areas that could be affected by the Federal action. The Federal action is the authorization or registration of pesticide use or uses as described on the label(s) of pesticide products containing a particular active ingredient. The action area is defined by the Endangered Species Act as, "all areas to be affected directly or indirectly by the Federal action and not merely the immediate are involved in the action" (50 CFR §402.2). Based on an analysis of the Federal action, the action area is defined by the actual and potential use of the pesticide and areas where that use could result in effects. Specific measures of ecological effect for the assessed species that define the action area include any direct and indirect toxic effect to the assessed species and any potential modification of its critical habitat, including reduction in survival, growth, and fecundity as well as the full suite of sublethal effects available in the effects literature. It is recognized that the overall action area for the national registration of PCNB is likely to encompass considerable portions of the United States based on the large array of agricultural and non-agricultural uses. However, the scope of this assessment limits consideration of the overall action area to those portions that may be applicable to the protection of the BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and their designated critical habitat within California. For this assessment, the entire state of California is considered the action area. The purpose of defining the action area as the entire state of California is to ensure that the initial area of consideration encompasses all areas where the pesticide may be used now and in the future, including the potential for off-site transport via spray drift and downstream dilution that could influence the San Francisco Bay Species. Additionally, the concept of a state-wide action area takes into account the potential for direct and indirect effects and any potential modification to critical habitat based on ecological effect measures associated with reduction in survival, growth, and reproduction, as well as the full suite of sublethal effects available in the effects literature.

It is important to note that the state-wide action area does not imply that direct and/or indirect effects and/or critical habitat modification are expected to or are likely to occur over the full extent of the action area, but rather to identify all areas that may potentially be affected by the action. The Agency uses more rigorous analysis including consideration of available land cover data, toxicity data, and exposure information to determine areas where BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and designated critical habitat may be affected or modified via endpoints associated with reduced survival, growth, or reproduction.

2.7.2. LAA Effects Determination Area

A stepwise approach is used to define the Likely to Adversely Affect (LAA) Effects Determination Area. An LAA effects determination applies to those areas where it is expected that the pesticide's use will directly or indirectly affect the species and/or modify its designated critical habitat using EFED's standard assessment procedures (see Attachment I) and effects endpoints related to survival, growth, and reproduction. This is the area where the "Potential Area of LAA Effects" (initial area of concern + drift distance or downstream dilution distance) overlaps with the range and/or designated critical habitat for the species being assessed. If there is no overlap between the potential area of LAA effects and the habitat or occurrence areas, a no effect determination is made. The first step in defining the LAA Effects Determination Area is to understand the federal action. The federal action is defined by the currently labeled uses for PCNB. An analysis of labeled uses and review of available product labels was completed. Some of the currently labeled uses are not specified for use in California (are restricted to Georgia) and are excluded from this assessment. In addition, a distinction has been made between food use crops and those that are non-food/non-agricultural uses. For those uses relevant to the assessed species, the analysis indicates that, for PCNB, the following agricultural uses are considered as part of the federal action evaluated in this assessment: cole crops, golf course GTF, cotton, potatoes and ornamental bulbs (dip/soak only).

Following a determination of the assessed uses, an evaluation of the potential "footprint" of PCNB use patterns (*i.e.*, the area where pesticide application may occur) is determined. This "footprint" represents the initial area of concern, based on an analysis of available land cover data for the state of California. The initial area of concern is defined as all land cover types and the stream reaches within the land cover areas that represent the labeled uses described above. For PCNB, this includes cultivated cropland and developed land classes.

Once the initial area of concern is defined, the next step is to define the potential boundaries of the Potential Area of LAA Effects by determining the extent of offsite transport via spray drift and runoff where exposure of one or more taxonomic groups to the pesticide will result in exceedances of the listed species LOCs. Thus, for PCNB, which is subject to long-range transport, the entire state of California is considered the Potential Area of LAA Effects.

2.8. Assessment Endpoints and Measures of Ecological Effect

A complete discussion of all the toxicity data available for this risk assessment, including resulting measures of ecological effect selected for each taxonomic group of concern, is included in Section 4 of this document. **Table 2-8** identifies the taxa used to assess the potential for direct and indirect effects from the uses of PCNB for each listed species assessed here. The specific assessment endpoints used to assess the potential for direct and indirect effects to each listed species are provided in **Table 2-9**.

Table 2-8. Taxa Used in the Analyses of Direct and Indirect Effects for the Assessed Listed Species.

Listed Species	Birds	Mammals	Terr. Plants	Terr. Inverts.	FW Fish	FW Inverts.	Estuarine/ Marine Fish	Estuarine/ Marine Inverts.	Aquatic Plants
San Francisco garter snake**	Direct Indirect (prey)	Indirect (prey/ habitat)	Indirect (habitat)	Indirect (prey)	Indirect (prey)	Indirect (prey)	n/a	n/a	Indirect (habitat)
California clapper rail**	Direct Indirect (prey)	Indirect (prey)	Indirect (food/ habitat)	Indirect (prey)	Indirect (prey)	Indirect (prey)	Indirect (prey)	Indirect (prey)	Indirect (food/ habitat)
Bay checkerspot butterfly	n/a	n/a	Indirect (food/ habitat) *	Direct	n/a	n/a	n/a	n/a	n/a
California tiger salamander	Direct	Indirect (prey/ habitat)	Indirect (habitat)	Indirect (prey)	Direct Indirect (prey)	Indirect (prey)	n/a	n/a	Indirect (food/ habitat)
Tidewater goby	n/a	n/a	Indirect (habitat)	n/a	Direct*** (IF MORE SENSITIVE THAN E/M FISH)	Indirect (prey)	Direct*** (IF MORE SENSITIVE THAN FW FISH)	Indirect (prey)	Indirect (habitat)
Delta smelt	n/a	n/a	Indirect (habitat)	n/a	Direct*** (IF MORE SENSITIVE THAN E/M FISH)	Indirect (prey)	Direct*** (IF MORE SENSITIVE THAN FW FISH)	Indirect (prey)	Indirect (food/ habitat)
California freshwater shrimp	n/a	n/a	Indirect (food/ habitat)	n/a	n/a	Direct Indirect (prey)	n/a	n/a	Indirect (food/ habitat)

Abbreviations: n/a = Not applicable; Terr. = Terrestrial; Invert. = Invertebrate; FW = Freshwater

* Obligate relationship

** Consumption of residues of PCNB in aquatic organisms may result in direct effects to the San Francisco Garter Snake and the Clapper Rail.

***The most sensitive fish species across freshwater and estuarine/marine environments is used to assess effects for these species because they may be found in freshwater and/or estuarine/marine environments.

Table 2-9. Taxa and Assessment Endpoints Used to Evaluate the Potential for Use of PCNB to Result in Direct and Indirect Effects to the Assessed Listed Species or Modification of Critical Habitat.

Taxa Used to Assess Direct and Indirect Effects to Assessed Species and/or Modification to Critical Habitat or Habitat	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
1. Freshwater Fish and Aquatic-Phase Amphibians	Direct Effect – -Tidewater Goby* -Delta Smelt* -California Tiger Salamander	Survival, growth, and reproduction of individuals via direct effects	1a. Most sensitive fish acute LC_{50} (guideline or ECOTOX) 1b. Most sensitive fish chronic NOAEC (guideline or ECOTOX) 1c. Most sensitive fish early-life stage NOAEC (guideline or ECOTOX
	Indirect Effect (prey) -SF Garter Snake -CA Clapper Rail	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.,* fish and aquatic-phase amphibians)	
2. Freshwater Invertebrates	Direct Effect – -CA FW Shrimp	Survival, growth, and reproduction of individuals via direct effects	2a. Most sensitive freshwater invertebrate EC_{50} (guideline or ECOTOX) 2b. Most sensitive freshwater invertebrate chronic NOAEC (guideline or ECOTOX)
	Indirect Effect (prey) -CA FW shrimp -SF Garter Snake -CA Clapper Rail -CA Tiger Salamander -Tidewater Goby -Delta Smelt	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.,* freshwater invertebrates)	
3. Estuarine/Marine Fish	Direct Effect – -Tidewater Goby* -Delta Smelt*	Survival, growth, and reproduction of individuals via direct effects	3a. Most sensitive estuarine/marine fish EC_{50} (guideline or ECOTOX) 3b. Most sensitive estuarine/marine fish chronic NOAEC (guideline or ECOTOX)
	Indirect Effect (prey) -Clapper Rail	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.,* estuarine/marine fish)	
4. Estuarine/Marine Invertebrates	Indirect Effect (prey) -CA Clapper Rail -Tidewater Goby -Delta Smelt	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.,* estuarine/marine invertebrates)	4a. Most sensitive estuarine/marine invertebrate EC_{50} (guideline or ECOTOX) 4b. Most sensitive estuarine/marine invertebrate chronic NOAEC (guideline or ECOTOX)
5. Aquatic Plants[‡] (freshwater/marine)	Indirect Effect (food/habitat) -SF Garter Snake -CA Clapper Rail -CA Tiger Salamander -Tidewater Goby -Delta Smelt -CA FW Shrimp	Survival, growth, and reproduction of individuals or modification of critical habitat/habitat via indirect effects on habitat, cover, food supply, and/or primary productivity (*i.e.,* aquatic plant community)	5a. Vascular plant acute EC_{50} (duckweed guideline test or ECOTOX vascular plant) 5b. Non-vascular plant acute EC_{50} (freshwater algae or diatom, or ECOTOX non-vascular)
6. Birds	Direct Effect -SF Garter Snake[†]	Survival, growth, and reproduction of individuals	6a. Most sensitive bird[†] or terrestrial-phase amphibian acute LC_{50} or LD_{50}

Taxa Used to Assess Direct and Indirect Effects to Assessed Species and/or Modification to Critical Habitat or Habitat	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
	-CA Clapper Rail -CA Tiger Salamander[†]	via direct effects	(guideline or ECOTOX) 6b. Most sensitive bird[†] or terrestrial-phase amphibian chronic NOAEC (guideline or ECOTOX)
	<u>Indirect Effect (prey/rearing sites)</u> -SF Garter Snake -CA Clapper Rail	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (birds)	
7. Mammals	<u>Indirect Effect (prey/habitat from burrows/rearing sites)</u> -SF Garter Snake -CA Clapper Rail -CA Tiger Salamander	Survival, growth, and reproduction of individuals via direct effects	7a. Most sensitive laboratory mammalian acute LC_{50} or LD_{50} (guideline or ECOTOX) 7b. Most sensitive laboratory mammalian chronic NOAEC (guideline or ECOTOX)
		Survival, growth, and reproduction of individuals or modification of critical habitat/habitat via indirect effects on terrestrial prey (mammals) and/or burrows/rearing sites	
8. Terrestrial Invertebrates	<u>Direct Effect</u> -Bay Checkerspot Butterfly	Survival, growth, and reproduction of individuals via direct effects	8a. Most sensitive terrestrial invertebrate acute EC_{50} or LC_{50} (guideline or ECOTOX) 8b. Most sensitive terrestrial invertebrate chronic NOAEC (guideline or ECOTOX)
	<u>Indirect Effect (prey)</u> -SF Garter Snake -CA Clapper Rail -CA Tiger Salamander	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (terrestrial invertebrates)	
9. Terrestrial Plants[‡]	<u>Indirect Effect (food/habitat) (non-obligate relationship)</u> -SF Garter Snake -CA Clapper Rail -SF Garter Snake -CA Tiger Salamander -Tidewater Goby -Delta Smelt	Survival, growth, and reproduction of individuals or modification of critical habitat/habitat via indirect effects on food and habitat (*i.e.,* riparian and upland vegetation)	9a. Distribution of EC_{25} for monocots (seedling emergence, vegetative vigor, or ECOTOX 9b. Distribution of EC_{25} (EC_{05} or NOAEC for the BCB and the VELB) for dicots (seedling emergence, vegetative vigor, or ECOTOX)
	<u>Indirect Effect (food/habitat) (obligate relationship)</u> -Bay Checkerspot Butterfly		

Abbreviations: SF=San Francisco

* The most sensitive fish species across freshwater and estuarine/marine environments is used to assess effects for these species because they may be found in freshwater or estuarine/marine environments.

[†] Birds are used as a surrogate for terrestrial-phase amphibians and reptiles.

[‡] No relevant toxicity data are available for these groups for PCNB or its degradates

As previously discussed, designated critical habitat is assessed to evaluate actions related to the use of PCNB that may alter the PCEs of the assessed species' designated critical habitat. PCEs for the assessed species were previously described in Section 2.6. Actions that may modify critical habitat are those that alter the PCEs and jeopardize the continued existence of the assessed species. Therefore, these actions are identified as assessment endpoints. It should be noted that evaluation of PCEs as assessment endpoints is limited to those of a biological nature (*i.e.*, the biological resource requirements for the listed species associated with the critical habitat) and those for which PCNB effects data are available.

Assessment endpoints used to evaluate potential for direct and indirect effects are equivalent to the assessment endpoints used to evaluate potential effects to designated critical habitat. If a potential for direct or indirect effects is found, then there is also a potential for effects to critical habitat. Some components of these PCEs are associated with physical abiotic features (*e.g.*, presence and/or depth of a water body, or distance between two sites), which are not expected to be measurably altered by use of pesticides.

2.9. Conceptual Model

2.9.1. Risk Hypotheses

Risk hypotheses are specific assumptions about potential adverse effects (*i.e.,* changes in assessment endpoints) and may be based on theory and logic, empirical data, mathematical models, or probability models (USEPA, 1998). For this assessment, the risk is stressor-linked, where the stressor is the release of PCNB and its degradates to the environment. The following risk hypotheses are presumed in this assessment:

The labeled use of PCNB within the action area may:

- directly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG by causing mortality or by adversely affecting growth or fecundity;
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of food supply;
- indirectly affect CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of the aquatic plant community in the species' current range, thus affecting primary productivity and/or cover;
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of the terrestrial plant community in the species' current range;
- indirectly affect CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing aquatic habitat in their current range (via modification of water quality parameters, habitat morphology, and/or sedimentation);
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing terrestrial habitat in their current range (via reduction in small burrowing mammals leading to reduction in underground refugia/cover).

2.9.2. Diagram

The conceptual model is a graphic representation of the structure of the risk assessment. It specifies the PCNB release mechanisms, biological receptor types, and effects endpoints of potential concern. The aquatic and terrestrial conceptual models for BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and associated PCE components of designated critical habitat are shown in **Figure 2-9** and **Figure 2-10**, respectively. Although the conceptual models for direct/indirect effects and modification of designated critical habitat PCEs are shown on the same diagrams, the potential for direct/indirect effects and modification of PCEs are evaluated separately in this assessment. Exposure routes shown in dashed lines are not quantitatively considered because the contribution of those potential exposure routes to potential risks to BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and modification to designated critical habitat is expected to be negligible.

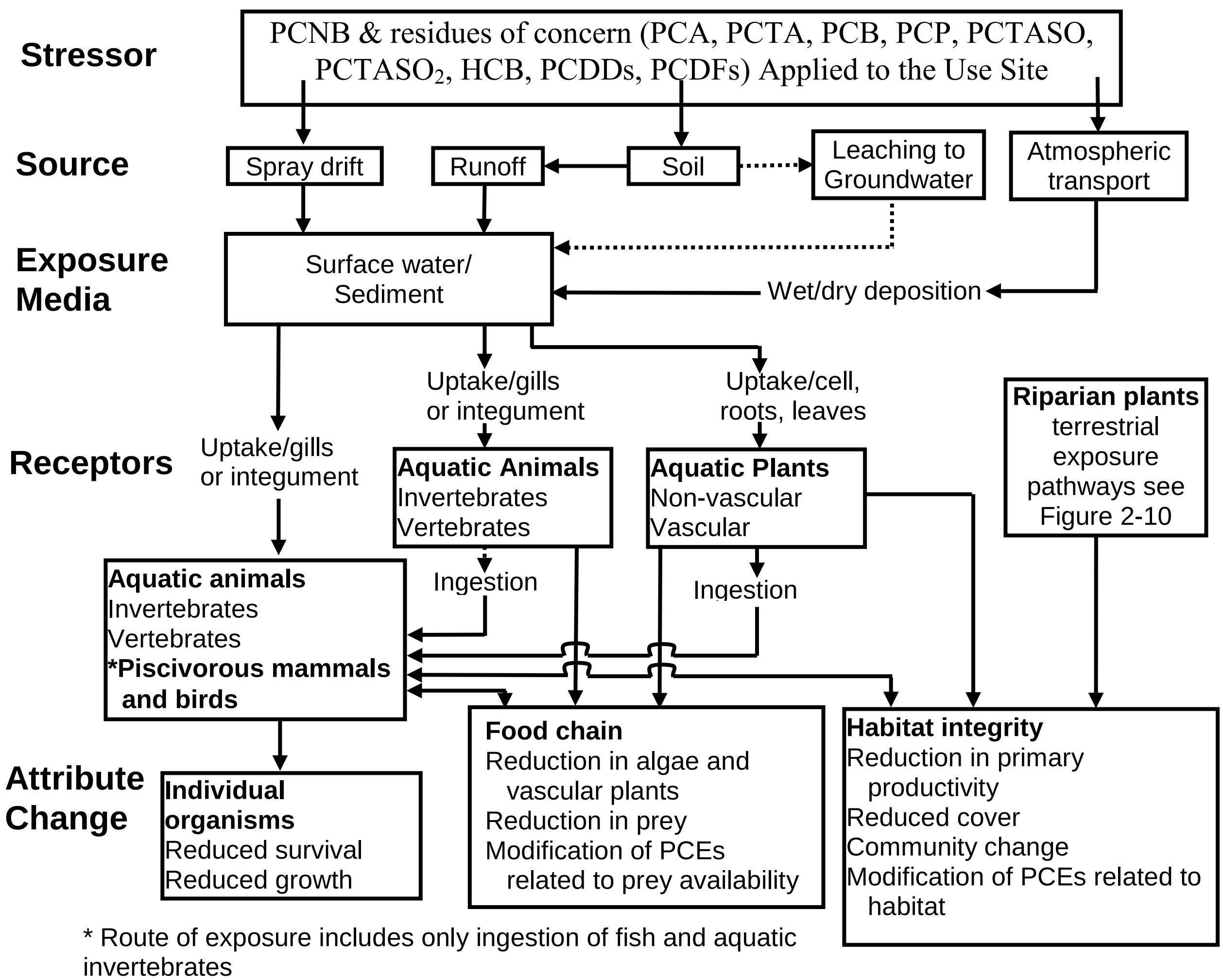


Figure 2-9. Conceptual Model Depicting Stressors (PCNB and its residues of concern), Exposure Pathways, and Potential Effects to Aquatic Organisms from the Use of PCNB.
Dotted lines indicate exposure pathways that have a low likelihood of contributing to ecological risk.

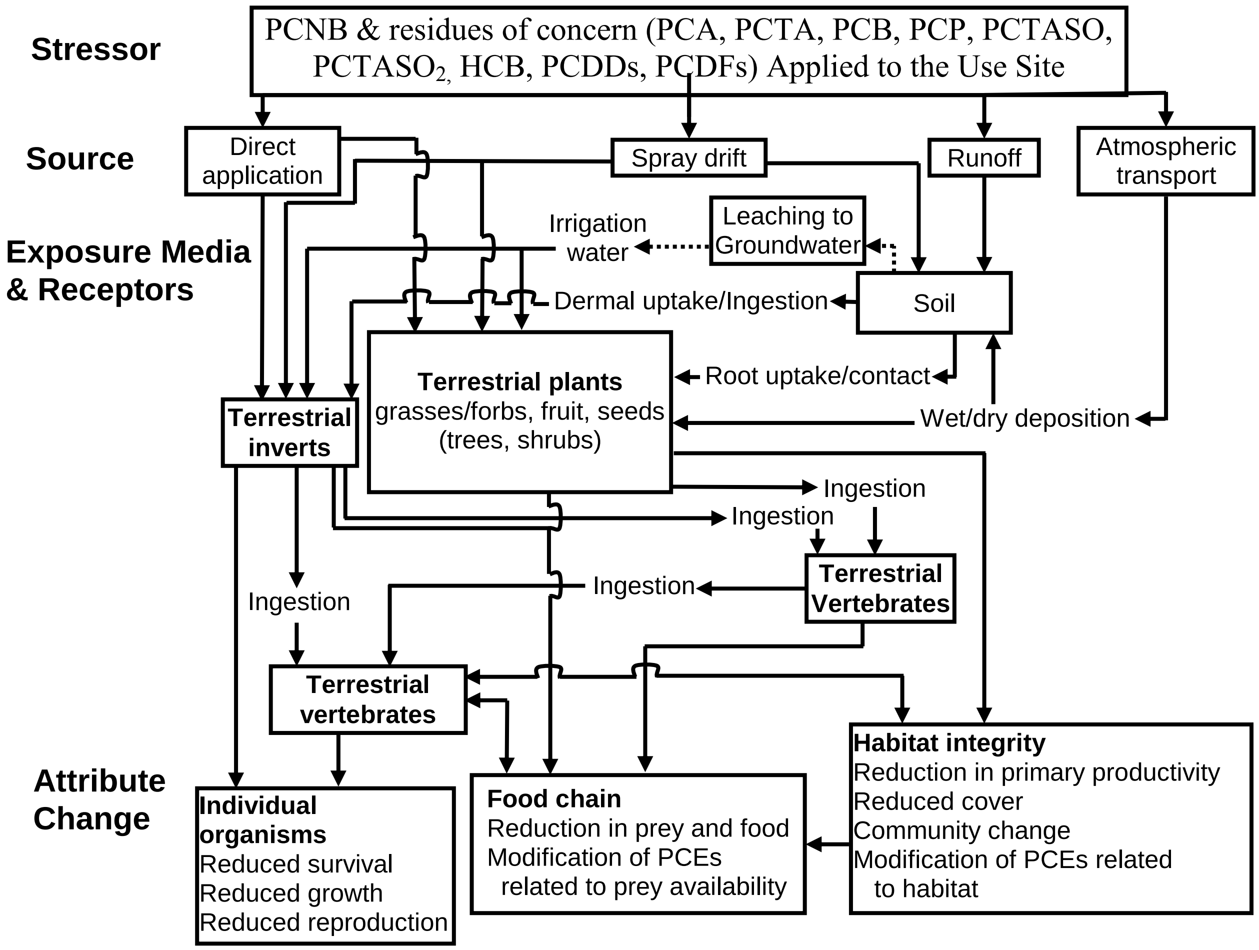


Figure 2-10. Conceptual model depicting stressors, exposure pathways, and potential effects to terrestrial organisms from the use of PCNB.

2.10. Analysis Plan

In order to address the risk hypothesis, the potential for direct and indirect effects to the assessed species, prey items, and habitat is estimated based on a taxon-level approach. In the following sections, the use, environmental fate, and ecological effects of PCNB are characterized and integrated to assess the risks. This is accomplished using a risk quotient (ratio of exposure concentration to effects concentration) approach. Although risk is often defined as the likelihood and magnitude of adverse ecological effects, the risk quotient-based approach does not provide a quantitative estimate of likelihood and/or magnitude of an adverse effect. However, as outlined in the Overview Document (USEPA, 2004), the likelihood of effects to individual organisms from particular uses of PCNB is estimated using the probit dose-response slope and either the level of concern (discussed below) or actual calculated risk quotient value.

Descriptions of routine procedures for evaluating risk to the San Francisco Bay Species are provided in Attachment I.

2.10.1. Measures of Exposure

The environmental fate properties of PCNB along with available monitoring data indicate that water and sediment runoff, and volatilization are the principle potential transport mechanisms of PCNB to the aquatic and terrestrial habitats. Spray drift will also be a potentially significant transport mechanism for the foliar use on golf course turf (GTF). Long-range transport (via atmospheric transport) and bioaccumulation are other potential pathways of exposure. In this assessment, transport of PCNB through runoff is considered in deriving quantitative estimates of PCNB exposure to BCB, CTS (all DPS), DS, CCR, CFWS, SFGS, TG, and their prey and habitats. Based on chemical properties and the results of laboratory studies, groundwater is not considered a likely route of exposure for PCNB.

Measures of exposure are based on aquatic and terrestrial models that predict estimated environmental concentrations (EECs) of PCNB using maximum labeled application rates and methods of application. The models used to predict aquatic EECs are the Pesticide Root Zone Model coupled with the Exposure Analysis Model System (PRZM/EXAMS). To estimate exposures to terrestrial species resulting from uses involving PCNB applications, the T-REX model is used for foliar and granular uses. The T-HERPS model is used to allow for further characterization of dietary exposures of reptiles and terrestrial-phase amphibians relative to birds. KABAM (v.1.0) is used to estimate potential bioaccumulation of PCNB residues in an aquatic food web and subsequent risks these residues pose to organisms consuming aquatic species. These models are parameterized using relevant reviewed registrant-submitted environmental fate data. More information on these models is available in Attachment I.

2.10.2. Measures of Effect

Data identified in Section 2.8 are used as measures of effect for direct and indirect effects. Data were obtained from registrant submitted studies or from literature studies identified by ECOTOX. More information on the ECOTOXicology (ECOTOX) database and how toxicological data is used in assessments is available in Attachment I.

2.10.3. Integration of Exposure and Effects

Risk characterization is the integration of exposure and ecological effects characterization to determine the potential ecological risk from agricultural and non-agricultural uses of PCNB, and the likelihood of direct and indirect effects to the assessed species in aquatic and terrestrial habitats. The exposure and toxicity effects data are integrated in order to evaluate the risks of adverse ecological effects on non-target species. The risk quotient (RQ) method is used to compare exposure and measured toxicity values. EECs are divided by acute and chronic toxicity values. The resulting RQs are then compared to the Agency's LOCs (USEPA, 2004) (see Appendix C for a description of the LOCs used in this assessment for each group of organisms). More information on standard assessment procedures is available in Attachment I.

2.10.4. Data Gaps

The environmental fate and ecological risk assessment chapter (USEPA, 2005) written in support of the Reregistration Eligibility Decision (RED) for PCNB characterized PCNB as a persistent, moderately volatile compound that would be immobile in most soils but may have slight to even moderate mobility in coarser (sandy) soils, particularly those that are low in organic matter. The current risk hypothesis for PCNB suggests the chemical and its degradates and contaminants are persistent (P), bioaccumulative (B) and toxic (T); as such, PCNB, viewed collectively with its degradates, has properties consistent with chemicals that are termed PBTs. The properties of the PBTs and of PCNB as well, lend themselves to long-range atmospheric transport leading to potential exposure of nontarget organisms distant from registered use sites and resultant accumulation and magnification of residues in food chains. However, because of the variability in available data and in some cases because of the lack of acceptable data, there is uncertainty regarding the extent to which PCNB and its degradates [that retain toxicity] are indeed persistent and prone to bioaccumulation and atmospheric transport. Additionally, there is uncertainty regarding the extent to which both the parent compound and its degradates are toxic to a broader range of taxa than those for which there are currently available data. From the perspective of reducing this uncertainty, the following data gaps have been identified:

- Data on the photodegradation of the degradate PCA are not available. Because of its stability in the environment and because it is the primary persistent degradate, an aqueous photolysis study of PCA is necessary. More comprehensive information on the environmental fate behavior of PCA would facilitate separate modeling estimates for this main degradate, which would more accurately characterize the fate of PCNB in aquatic and terrestrial habitats. In the absence of these data, it is assumed that, in general, PCNB would degrade to PCA in soil and that the PCA would then remain stable to further degradation in both the aquatic and terrestrial environments.

- Data on photodegradation in air are not available. There is uncertainty regarding the fate of the parent compound and degradates once PCNB has volatilized. Based on air monitoring data demonstrating the presence of PCNB, uncertainty regarding the extent to which the compound would be subject to photodegradation in the atmosphere (when present in water vapor sorbed to particulates) or to atmospheric photooxidation, and QSAR estimates of atmospheric photooxidation rates, the parent compound is assumed to be stable and subject to atmospheric transport. Given that PCA is the major degradate remaining after microbial degradation, and is a volatile compound, photodegradation in air data are needed for both PCNB and PCA to more accurately characterize the extent to which PCNB and/or PCA will persist in the atmospheric compartment and be available for long-range transport. In the absence of such data, it is assumed that PCNB and PCA are likely to volatilize from applications where it is not incorporated into soil, are stable in the atmosphere, and subject to long-range atmospheric transport. These are conservative assumptions, which could result in overstating the potential for long-range transport of PCNB and PCA.

- There are no aquatic metabolism guideline data (aerobic or anaerobic) available, and there are limited soil metabolism data available for PCNB. There is uncertainty regarding the extent to which PCNB is subject to biotic degradation in both the terrestrial and aquatic environments. The RED characterized the parent as likely to biodegrade slowly under aerobic conditions, but noted that PCNB was metabolized more rapidly under anaerobic soil conditions (USEPA, 2006). There is uncertainty regarding the potential formation of PCP and pentachlorobenzene (PCB), although the later is known to be a manufacturing impurity along with hexachlorobenzene (HCB), formed during the production of PCNB. Because the available studies do not fully characterize the formation and decline of degradation products, aerobic and anaerobic soil and aquatic metabolism studies are still needed. In the absence of these data, EFED will continue to assume that PCA, PCTA and PCB are major degradates which are stable to further degradation and contribute to total persistent residues of PCNB. This assumption could result in overstating the persistence of PCNB residues and risk to aquatic and terrestrial organisms.

- Data on the foliar dissipation half-life of PCNB are not available and therefore terrestrial exposure assessments have relied on the default value of 35 days. While volatility may substantially reduce the amount of PCNB residues following foliar applications, there are no data submitted to determine a more appropriate value for use in terrestrial exposure assessments.

- Data on the toxicity of PCNB to sediment-dwelling invertebrates are not available. The likely presence of PCNB in benthic sediments could serve as a route of entry into aquatic food chains.

- Chronic data for PCNB on estuarine/marine fish and invertebrates are not available. Given the use of PCNB on golf courses, the proximity of some golf courses to coastal areas, and the use of PCNB on crops, *e.g*. cole crops grown in coastal locations, coupled with the likely persistence of the compound, there is a potential for chronic exposure in estuarine environments.

- Data on the toxicity of PCNB to terrestrial, semi-aquatic, or aquatic plants are not available; the only incident reported in the Agency's Ecological Incident Information System (EIIS) for PCNB involves plants suggesting that plants may be sensitive to PCNB. Additionally, the propensity for aquatic plants to bioconcentrate PCNB underscores the need for better understanding the effects of PCNB on plants.

- Open literature data indicate that some of the metabolites (pentachloroaniline, PCA; pentachlorothioanisole, PCTA; pentachlorobenzene, PCB; PCTA sulfoxide; PCTA sulfone; and pentachlorophenol, PCP) of PCNB are toxic. However, OPP has not received specific toxicity data from the registrant(s) with which to evaluate the toxicity of those metabolites. This assessment relies on open literature studies identified through EPA's ECOTOXicology (ECOTOX) database[5] for toxicity data related to the degradates PCA and PCB.

[5] http://cfpub.epa.gov/ecotox/

- No data are available in the open literature on aquatic- or terrestrial-phase amphibians, reptiles, or lepidopterans, and EPA does not require registrants to submit data on these species. In this assessment, fish are used as surrogates for aquatic-phase amphibians, birds are used as surrogates for terrestrial-phase amphibians and reptiles, and honey bees are used as surrogates for lepidopterans.

3. Exposure Assessment

PCNB is formulated as a flowable and a granular. Aerial, chemigation, and hand-held application are prohibited. Turfcide® 4F may be applied by bulb soak or ground-boom application only, and Turfcide® 10% Granular may be applied by tractor-drawn spreaders only. Except for turf and bulbs, all applications are soil applied and incorporated at the time of planting. All these are in-furrow applications, except for cole crops, which may be band-row applied or broadcast and then disked in. Bulbs are soaked in a solution of PCNB prior to planting. All applications to golf course turf must be followed by ¼-inch of irrigation water. Applications to ornamental bulbs are currently limited to bulb soak applications only.

For aquatic exposure, risks from ground boom application of the flowable formulation are considered in this assessment because they are expected to result in the highest off-target levels of PCNB due to generally higher spray drift levels than application of the granular formulation. For the turf use, however, application of the granular by ground-spreader is also assessed for aquatic exposure because that use has a higher application rate for turf than the one associated with the flowable formulation.

For terrestrial exposure, both granular and foliar uses of PCNB are assessed separately for all uses except potatoes, which only has an approved foliar use in California.

3.1. Label Application Rates and Intervals

PCNB labels may be categorized into two types: a label for manufacturing uses (including technical grade PCNB and its formulated products) and end-use products. While the technical product (Reg. # 5481-197), which contains PCNB of high purity, is not used directly in the environment, it is used to make formulated products (Turfcide® 4F, #5481-8992; and Turfcide® 10% Granular, # 5481-8988) which can be applied in specific areas to control soil borne diseases such as club root, brown patch, melting out/leaf spot/dollar spot, gray snow mold, pink snow mold, root rot/stem rot/bulb rot, bulb rot/crown rot, black rot, and stem canker/black scurf. The formulated product labels legally limit PCNB's potential use to only those sites that are specified on the labels.

Section 2.4.6, Use Characterization, includes a description of pending label change requests and ongoing mitigations that that may result in changes to the PCNB label.

Currently registered agricultural and non-agricultural uses of PCNB within California include cole crops (Brussels sprouts, broccoli, cabbage, cauliflower), cotton, potato, ornamental bulbs (dip/soak only), and turf (golf course greens, tees, and fairways only). For turf, PCNB is only registered in CA for snow mold control on golf courses. Other labeled uses on golf course turf (i.e., dollar spot, melting out/leaf spot) are prohibited in California, so were not assessed. The uses being assessed are summarized in **Table 3-1**.

Table 3-1. PCNB Uses, Scenarios, and Application Information for Use in California.

Scenario	Uses Represented by Scenario	Application Method/ Formulation	Application Rate (lbs a.i./acre)	Maximum Number of Applications	Application Interval
CA Cole Crops	Brussels sprouts, cabbage, broccoli, cauliflower	Ground spray/ flowable	22.5	1 (at planting)	n/a
CA Cotton w/irrig.	cotton	Ground spray/ flowable	2	1 (at planting)	n/a
CA Potato	potato	Ground spray/ flowable	5	1 (at planting)	n/a
CA Turf	Turf on golf course greens, tees, fairways	foliar spray/ flowable	33	2	28
CA Turf	Turf on golf course greens, tees, fairways	broadcast by ground spreader/ granular	43.56	2	28
Bulbs	No Scenario Available	Dip (bulb soak; flowable)	Not Specified on Label.	1 (pre-planting)	n/a

Uses assessed based on memorandum from Pesticide Re-evaluation Division (PRD) dated 5/15/2012 and EFED Label Data report and associated Label Use Information Reports prepared on 4/3/2012.

Use on ornamental bulbs is not assessed due to a lack of definitive information. BEAD attempted to estimate a per-acre rate for the use of PCNB on ornamental bulbs. Using information from the Easter Lily Research Foundation through the Washington State University Tri-Cities Extension (2012) on planting density per row foot and dip suspension rates from the label, BEAD determined that

> "*If calculated according to the plant density, depending on the stage of growth up to 80 plants per row foot (1,161,600 plants per acre) would require 141 lb PCNB per acre (1,161,600 plants per acre/8222 bulbs per lb PCNB). The PCNB label lists dip suspension rates of 1.5 lb per 3.2 gallons of water for control of bulb/crown rot (Sclerotium rolfsii) and bulb and stem rot (Sclerotinia sclerotiorum) on iris, hyacinth, narcissus, and tulips, which (if calculated to an acre of plants) would result in the use of several hundred pounds of active ingredient.*"

Because the bulb dip use results in *de facto* incorporation of the pesticide, it should be less available for surface runoff relative to a foliar use. However, the lack of information on the actual rates of applied pesticide preclude an exposure assessment for this use.

3.2. Aquatic Exposure Assessment

3.2.1. Modeling Approach

The EECs (Estimated Environmental Concentrations) for both surface water and sediment are calculated using the EPA Tier II PRZM (Pesticide Root Zone Model) and EXAMS (Exposure Analysis Modeling System) with the EFED Standard Pond environment. PRZM is used to simulate pesticide transport as a result of runoff and erosion from an agricultural field, and EXAMS estimates environmental fate and transport of pesticides in surface water. Aquatic exposure is modeled for PCNB total residues of concern (defined here to include the parent, PCA, PCTA, PCB, PCP, PCTASO, $PCTASO_2$ and/or HCB, including volatiles).

The most recent PRZM/EXAMS linkage program (PE5, PE Version 5, dated Nov. 15, 2006) was used for all surface water simulations. Linked crop-specific scenarios and meteorological data were used to estimate exposure resulting from use on crops and turf (golf course greens, tees and fairways). The use on ornamental bulbs could not be modeled due to insufficient information on application rates and because a bulb scenario is not available. Because the bulb use is limited to bulb dip only, and uptake efficiency rates and field sizes are not known, BEAD was not able to calculate a reasonable estimated use rate for the field.

Use-specific management practices for all of the assessed uses of PCNB were used for modeling, including application rates, number of applications per year, and the first application date for each use. All applications assessed were at planting only with the exception of turf. The initial timing of turf treatment was based on the start of favorable conditions for the propagation of snow mold in northern California (*i.e.*, late fall to early winter). The dates of application were developed based on several sources of information including label instructions, data provided by BEAD, emergence dates in the modeling scenarios, and crop profiles maintained by the USDA.

More detail on the crop profiles and the previous assessments may be found at:
http://www.ipmcenters.org/CropProfiles/

To determine aquatic EECs, aquatic exposure modeling was conducted using use parameters specific to the flowable formulation for all uses. Because the use rate for turf is higher for the granular formulation (2 applications of 43.56 lbs ai/A) than the flowable formulation (2 apps. of 33 lb ai/A), both use patterns were modeled for turf.

Because PCNB use on turf is strictly on golf courses, estimated surface water concentrations were post-processed to account for pesticide use on less than the total turf acreage in the modeled watershed. Golf course facilities consist of separate playing areas that are classified as tees, greens, practice green, fairways, driving range, and roughs, in addition to “unmanaged grounds” where lakes, ponds, out-of-play areas, conservation areas, and buildings are located. Management practices and intensity vary with the type of playing area, and it is common for pesticides to be applied only to tees, greens and fairways. Because the label restricts PCNB use on golf course turf to tees, greens and fairways, and excludes roughs, a “Golf Course Adjustment Factor” (GCAF) is applied to the EEC’s determined using the Tier II model PRZM/EXAMS. However, the GCAF is only applicable to golf course use scenarios, and cannot be used to modify estimated surface water concentrations associated with sod farms, right-of-ways, or recreational fields turf uses that may be added to the label via the pending PRIA action discussed previously.

To refine the EECs for PCNB for golf course turf use on tees, greens, and fairways, the EECs were multiplied by 0.34 based on the premise that tees and greens account for a total of 5% of total managed golf course turf and fairways account for 29%. More information on the GCAF and how the values were determined may be found at: http://www.epa.gov/oppefed1/models/water/golf_course_adjustment_factors.htm.

3.2.2. Model Inputs

The appropriate PRZM, and EXAMS input parameters for PCNB were selected from the environmental fate data submitted by the registrant and in accordance with US EPA-OPP EFED water model parameter selection guidelines, *Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides. Version 2.1*, October 22, 2009 and *PE5 User's Manual. (P)RZM (E)XAMS Model Shell, Version (5)*, November 15, 2006. Input parameters can be grouped by physical-chemical properties and other environmental fate data, application information, and use scenarios. Physical and chemical properties relevant to assess the behavior of PCNB and related compounds in the environment are presented in **Table 2-1** and **Table 2-2** and application information from the label in **Table 2-3** and **Table 3-1**. The input parameters for PRZM and EXAMS are in **Table 3-2**. Appendix D contains example model output files and tables showing the data used to calculate input values.

Table 3-2. Summary of PRZM/EXAMS Environmental Fate Data Used for Aquatic Exposure Inputs for PCNB Endangered Species Assessment[1].

Fate Property	Value (unit)	MRID (or source) & Comments
Molecular Weight	295.3 g/mole	USEPA, 2005.
Henry’s constant	9.980E-005 atm-m3/mole	Calculated from water solub. & vapor pressure
Vapor Pressure	1.13 x 10^{-4} Torr @ 25°C	USEPA, 2005.
Solubility in Water	0.44 mg/L @ 25°C	USEPA, 2005.
Photolysis in Water	1.8	MRIDs 42606201, 42606202, 42336201
Aerobic Soil Metabolism Half-lives	1124 days (parent plus degradates)	MRIDs 42911902, 41384501, 41713202, 42112801; represents the 90th percentile of the upper confidence bound on the mean of 2 half-life values. Does not include PCDD or PCDF residues.
Hydrolysis Half-lives	Stable	MRIDs 40865301, 40972601

Fate Property	**Value (unit)**	**MRID (or source) & Comments**
Aerobic Aquatic Metabolism Half-life (water column)	2248 days (parent plus degradates)	Input value is 2X the aerobic soil metabolism half-life input value . Does not include PCDD or PCDF residues.
Anaerobic Aquatic Metabolism Half-life (benthic)	2004 days (parent plus degradates)	Input value is two times the anaerobic soil metabolism half-life value (as in input parameter guidance); calculated using source data from MRID 41384301. Does not include PCDD or PCDF residues.
Organic-carbon water partition coefficient (K_{OC}, L/kg OC)	6470	41648201 Mean of 4 values for parent.
Application date (day/month)	Cole crops: 15/8 Cotton: 20/4 Potatoes: 20/5 Turf: 1/11 (foliar spray) Turf: 1/11 (granular)	All at-plant applications except turf, which is treated when conditions become favorable for snow mold.
Application rate and frequency	Cole crops: 25.2 kg/ha (22.5 lb/A) Cotton: 2.24 kg/ha (2 lb/A) Potatoes: 5.6 kg/ha (5 lb/A) Turf: 37.0 kg/ha (33 lb/A x 2 applic.; foliar spray) Turf: 48.79 kg/ha (43.56 lb/A x 2 applic.; granular)	All are single applications except for turf, which is two applications per year.
Application intervals	Not applicable except for turf. Turf: 28 days (foliar spray) Turf: 28 days (granular)	All applications are at-plant or immediately pre-plant except for turf.
Chemical Application Method (CAM)	Cole crops: 4 (banded) Cotton: 1 (in furrow and on surface) Potatoes: 4 (in furrow) Turf: 2 (foliar spray) Turf: 1 (broadcast granular)	
Application Efficiency	Cole crops: 0.99 Cotton: 0.99 Potatoes: 0.99 Turf: 0.99 (foliar spray) Turf: 1.0 (granular)	
Spray Drift Fraction	Cole crops: 0.01 Cotton: 0.01 Potatoes: 0.01 Turf: 0.01 (foliar spray) Turf: 0 (granular)	
Incorporation Depth	Cole crops: 10 cm Cotton: 0 Potatoes: 10 cm Turf: 0 (foliar spray) Turf: 0 (granular)	

Fate Property	Value (unit)	MRID (or source) & Comments
Post-harvest foliar pesticide disposition (IPSCND)	Cole crops: 1 Cotton: 1 Potatoes: 1 Turf: 3 (foliar spray) Turf: 1 (granular)	

[1]Inputs determined in accordance with EFED "*Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides. Version 2.1"* dated October 22, 2009.

3.2.3. Results

The aquatic EECs for the various scenarios and application practices are listed in **Table 3-3**. The output/example output from PRZM-EXAMS is provided in Appendix D. The use of PCNB on cole crops resulted in the highest aquatic EECs, with respective peak, 21-day average and 60-day average values of 31.0, 12.1, and 8.7 ug/L. Turf uses resulted in similar EECs for water for the two use patterns (*i.e.*, foliar spray of flowable and groundspreader broadcast of granular) despite the fact that the flowable formulation is only applied at 75% of the granular formulation rate. Turf EECs for water were approximately half of the water EECs for cole crops, while cotton and potato EECs (which were similar) were around 11% or less of the cole crop EECs. The 60-day average EECs were approximately 20-28% of the peak EECs across all uses, indicating the pesticide will not persist in the water over time, but will likely volatilize or dissipate to the sediment phase.

Relative to the overlying water EECs, similar patterns occurred among sediment EECs between uses, with the highest associated with cole crops and the lowest associated with cotton and potato uses, which had similar sediment EECs. For turf use, as with water EECs, the sediment EECs were similar for the two use patterns (*i.e.*, foliar spray of flowable and groundspreader broadcast of granular) despite different application rates. However, unlike the overlying water EECs, the benthic sediment EECs did not show much decline from peak values to the 60-day average values, indicating that sediment loads are estimated to remain relatively constant over time.

Table 3-3. Aquatic EECs for Surface Water (µg/L) and Sediment (µg/g) for PCNB Uses in California.

Scenario (Application Method/ Formulation)	**Crops/Uses Represented**	**Applic. Rate**	**Date of First Applic.**	**No. of Applic.**	**Applic. Interval (days)**	**Peak EEC** (µg/L or µg/kg)	**21-day average EEC** (µg/L or µg/kg)	**60-day average EEC** (µg/L or µg/kg)
CA Cole Crops (banded/flowable)	Brussels sprouts, broccoli, cabbage, cauliflower	22.5 lb ai/A	Jan. 1	1	n/a	Water: 31.0 Sediment: 6.0	Water: 12.1 Sediment: 6.0	Water: 8.7 Sediment: 5.8
CA Cotton w/irrigation (in-furrow and surface spray/ flowable)	Cotton	2 lb ai/A	Apr. 15	1	n/a	Water: 3.0 Sediment: 0.39	Water: 1.1 Sediment: 0.39	Water: 0.61 Sediment: 0.38

Scenario (Application Method/ Formulation)	Crops/Uses Represented	Applic. Rate	Date of First Applic.	No. of Applic.	Applic. Interval (days)	Peak EEC (µg/L or µg/kg)	21-day average EEC (µg/L or µg/kg)	60-day average EEC (µg/L or µg/kg)
CA Potato (in-furrow spray/ flowable)	Potato	5 lb ai/A	Jan. 1	1	n/a	Water: 3.5 Sediment: 0.45	Water: 1.3 Sediment: 0.44	Water: 0.74 Sediment: 0.41
CA Turf (foliar spray/flowable)	Turf on golf course greens, tees, fairways	33 lb/A	Nov. 1	2	28	Water: 19.5* Sediment: 4.5	Water: 7.1 Sediment: 4.4	Water: 5.0 Sediment: 4.3
CA Turf (broadcast by ground spreader/ granular)	Turf on golf course greens, tees, fairways	43.56 lb/A	Nov. 1	2	28	Water: 18.9[1] Sediment: 4.4	Water: 7.5 Sediment: 4.3	Water: 5.2 Sediment: 4.2

* Turf EECs were post-processed using the Golf Course Adjustment Factor of 0.34. Turf EECs for water prior to adjustment were 57.47, 20.95, 14.71 ug/L for foliar spray and 55.69, 22.07, and 15.25 ug/L for broadcast granular. Turf EECs for sediment prior to adjustment were 13.16, 13.01, and 12.60 ug/L for foliar spray and 12.94, 12.76, 12.42 ug/L for broadcast granular.

3.2.4. Existing Monitoring Data

A critical step in the process of characterizing EECs is comparing the modeled estimates with available surface water monitoring data. Included in this assessment are PCNB data from the USGS National Water Quality Assessment (NAWQA) program (http://water.usgs.gov/nawqa) and data from the California Department of Pesticide Regulation (CDPR). In addition, air monitoring data and fish tissue data for PCNB are summarized. Limited monitoring data were found for PCNB and its major degradates PCA and PCTA. Available monitoring data for the degradates PCB and PCP are not presented here as PCNB is not the single source for these compounds in the environment and the available monitoring data could not be linked to the use of PCNB.

3.2.4.a. USGS NAWQA Surface Water Data

PCNB is not one of the analytes monitored in surface water for the USGS NAWQA Surface Water Monitoring program, but is monitored in the surface water bed sediment. In 49 sites during the years 1995 – 1998, PCNB was detected at <50 ug/kg at all sites with the exception of four (Riverside, Orange and twice in San Bernadino), at which it was detected at <100 µg/kg. Specific data ranges were not reported. Data are available at http://infotrek.er.usgs.gov/nawqa_queries/jsp/swmaster.jsp.

3.2.4.b. USGS NAWQA Groundwater Data

PCNB is not one of the analytes monitored in the USGS NAWQA Groundwater Monitoring program. Based on information contained in the USEPA's Pesticides *in Ground Water Database, A Compilation of Monitoring Studies: 1971-1991, National Summary*, PCNB is not found in groundwater at significant levels or frequencies (USEPA, 1992a). In sampling of 1708 wells, only three detections of PCNB occurred, at a range of 0.008–0.275 µg/L. Sampled wells which did not contain measurable levels of PCNB included 459 wells in California. However, there is no related information available with regard to whether the monitoring sites corresponded with PCNB use sites or times of usage. In another groundwater monitoring study of 18 wells in three counties in California (July 1994–1995), PCNB was not detected (California EPA, 1995).

3.2.4.c. California Department of Pesticide Regulation (CDPR) Data

PCNB was not detected in drainage ditches in California which were monitored for its presence, with a level of quantitation of 0.1 ppm. CPDR data can be found at: http://www.cdpr.ca.gov/docs/emon/surfwtr/surfcont.htm

3.2.4.d. Atmospheric Monitoring Data

Monitoring studies reported in the literature that involved PCNB included studies conducted in Canada where PCNB was detected in air over Saskatchewan (SK). PCNB was detected in 10 of 11 samples (at 0.01-1.60 ng/m^3) collected during a 3-month period in the summer of 1994 over Regina, SK, and in 2 of 5 samples collected during that time over Waskesiu, SK (**Table 3-4**). The compound was not detected in any of 7 samples collected over Yellowknife, SK. The authors indicated that the registered uses of PCNB in Canada at that time (*i.e.,* ornamentals, ginseng and turf) were not "major crops" grown in that part of SK (Thompson *et al.,* 1997).

Table 3-4. Measured concentrations of PCNB in air samples collected at areas removed from application sites.

Location	Date	Measured concentrations (pg/m^3)	Citation
Regina, Saskatchewan	Summer 1994	10-1600	Thompson *et al.,* 1997
Waskesiu, Saskatchewan	Summer 1994	30-380	Thompson *et al.,* 1997
Mount Revelstoke national Park, British Columbia	2003-2004	6-17	Daly *et al.,* 2007
Yoho National park, British Columbia	2003-2004	2-5	Daly *et al.,* 2007
Banff National Park, Alberta	2003-2004	0.4-1.3	Daly *et al.,* 2007

In another study that was conducted in Canada, PCNB was detected in air samples collected from 3 national parks in 2003 and 2004. At Mount Revelstoke National park, measured concentrations ranged 6-17 pg/m^3. At Yoho National park, measured concentrations ranged from 2 to 5 pg/m^3. At Observation Peak in Banff National Park, measured concentrations of PCNB ranged from 0.4 to 1.3 pg/m^3 (Daly *et al.,* 2007). The detection of PCNB in the atmosphere over areas in which the pesticide was not used is evidence of the possibility for long-range transport of the compound.

In a study of air samples collected in 1994 over areas of non-farmed land near Larimore, ND, PCNB was detected in 1 of 7 samples, at a concentration of 0.0017 ng/m^3 (Hawthorne *et al.,* 1996).

In an air monitoring study of pesticides in Lompoc, California, PCNB was detected in ambient air at a maximum acute (24-hr) concentration of 48 ng/m^3, a maximum subchronic concentration of 18 ng/m^3, and a maximum chronic concentration of 8.5 ng/m^3 (CDPR 2003; http://www.cdpr.ca.gov/docs/dprdocs/lompoc/lompoc.htm).

It is noted that PCNB is a listed Hazardous Air Pollutant (HAP) for which standards must be set under the Clean Air Act (amended 1990). PCNB is not listed on the U.S. EPA's list of Persistent Organic Pollutants (POPs) as found at: http://www.epa.gov/oppfead1/international/pops.html.

3.2.4.e. Fish Tissue Concentration Data

OPP has identified two monitoring studies that include data for PCNB concentrations in fish. In the first study, USEPA, collected fish samples at 362 sites. PCNB was detected in 4 samples of fish, with all detections <50 µg/kg (USEPA, 1992b).

In the second study, data on PCNB concentrations in fish are available from the National Lake Fish Tissue Study (NLFTS), which was the first survey of fish contamination in lakes and reservoirs in the 48 contiguous states based on a probability survey design (http://www.epa.gov/waterscience/fish/study/, Stahl *et al.,* 2009, and Olsen *et al.,* 2009). This study included 268 of the chemicals with PBT properties, including PCNB. The USEPA, implemented the study in cooperation with states, tribal nations, and other federal agencies, with field collection occurring at 500 lakes and reservoirs over a four-year period (2000–2003). The sampled lakes and reservoirs were selected using a spatially balanced unequal probability survey design from 270,761 lake objects in USEPA's River Reach File Version 3 (RF3). In the NLFTS, there were a total of 1003 composite fish samples in the U.S. that were evaluated for PCNB. Of these samples, 90 (9%) contained detectable levels of PCNB, with a concentration range of 0.766-5.54 µg/kg. PCNB was detected in fish samples collected throughout the 4-year sampling period and throughout the continental US (including locations in: AL, AZ, CA, CO, FL, IL, IN, KS, ME, MI, MN, MS, MI, MT, NE, NV, NH, NJ, NY, OH, OK, OR, RI, TX, UT, VA, WA, WI, WY). Detections occurred in both predator and bottom-dwelling fish.

These monitoring data likely underestimate PCNB's residues of concern because the monitoring studies focused only on PCNB alone; fish tissues were not analyzed for PCNB's degradates. This is especially a concern given that the PCNB degradate PCTA has been identified as representing the majority of PCNB residues in the two registrant-submitted BCF studies with bluegill sunfish. Additionally, these monitoring data likely underestimate concentrations of PCNB since the fish sampled during this monitoring program were not targeted to coincide with either PCNB application areas (spatially) or application periods (temporally). As such, the residues measured in the fish samples likely underestimate residues in fish collected in close proximity to PCNB use sites.

3.2.5. Total Toxic Residues (TTR) Approach

A total toxic residues (TTR) approach will be used in this assessment to capture potential exposure of aquatic organisms to PCNB degradates and impurities. The TTR method involves summing residues observed in fate studies and then estimating degradation rates based on the total summed residues. The TTR degradation rates will then be used to estimate exposure in place of degradation rates for the parent alone. The resulting EECs are compared to the most sensitive toxicity endpoint from either parent or degradate data for each group of aquatic organisms.

3.3. Terrestrial Exposure Assessment

3.3.1. Exposure to Residues in Terrestrial Food Items: Model Scenarios

In this assessment, exposure of terrestrial organisms are considered for parent (PCNB) residues only, and does not include any modeling of degradate residues; therefore, a TTR approach is not taken for terrestrial organisms. EFED currently does not evaluate potential exposures to degradates in its standard risk assessment process.

T-REX (Version 1.4.1) is used to calculate dietary and dose-based EECs of PCNB for birds (including terrestrial-phase amphibians and reptiles), mammals, and terrestrial invertebrates. T-REX simulates a 1-year time period. T-HERPS is used as a refinement of dietary and dose-based EECs for snakes and amphibians when risk quotients from T-REX are higher than LOCs. T-HERPS is also set up to simulate a 1-year time period. For this assessment, spray and granular applications of PCNB are considered for all uses except potatoes (spray only). Terrestrial EECs are derived for the uses previously summarized in **Table 3-1**. Crops with similar use patterns (e.g., cole crops) are grouped together with one crop chosen as a surrogate for the group for the purpose of data presentation. Exposure estimates generated using T-REX and T-HERPS are for the parent alone.

Terrestrial EECs for foliar formulations of PCNB were derived for the use scenarios summarized in **Table 3-5**. Given that foliar dislodgeable residue data are not available for PCNB, a default foliar dissipation half-life of 35 days is used based on the work of Willis and McDowell (1987). Turf is the only use of PCNB affected by the foliar dissipation rate since all other uses allow only a single application. The impact of the assumption of the 35 day foliar dissipation rate on turf EECs and risk estimates is discussed in the section of the document addressing uncertainties (6.2.1). Use specific input values, including number of applications, application rate, foliar half-life and application interval are provided in **Table 3-5**. An example output from T-REX and T-HERPS is available in Appendix E.

Table 3-5. Input Parameters for Foliar Applications Used to Derive Terrestrial EECs for PCNB with T-REX and T-HERPS.

Use (Application method)	Application Rate (lbs a.i./A)	Number of Applications	Application Interval	Foliar Dissipation Half-Life
Cole crops* (ground spray; banded)	22.5	1	N/A	35 days
Cotton (ground spray; in-furrow and on surface)	2	1	N/A	
Potatoes	5	1	N/A	
Turf (foliar spray)	33	2	28 days	

N/A = Not applicable

* Represents all cole crops but information presented is based on one crop of Brussels sprouts grown annually

Organisms consume a variety of dietary items and may exist in a variety of sizes at different life stages. T-REX estimates exposure for the following dietary items: short grass, tall grass, broadleaf plants/small insects, and fruits/pods/seeds/large insects, and seeds for granivores. Birds, including the CCR, and mammals, consume all of these items. The size classes of birds represented in T-REX are small (20 g), medium (100 g), and large (1000 g). The size classes for mammals are small (15 g), medium (35 g), and large (1000 g). EECs are calculated for the most sensitive dietary item and size class for birds (surrogate for amphibians and reptiles) and mammals. For birds and mammals, the most sensitive EECs are for the smallest size class consuming short grass. The percentages of the EECs for the different dietary items are discussed in the discussion on uncertainties (Section 6.1.1.a).

For foliar applications of liquid formulations, T-HERPS estimates exposure for the following dietary items: broadleaf plants/small insects, fruits/pods/seeds/large insects, small herbivore mammals, small insectivore mammals, and small amphibians. Snakes and amphibians may consume all of these items. The default size classes of amphibians represented in T-HERPS are small (2 g), medium (20 g), and large (200 g). The default vertebrate prey size that the medium and large amphibians can consume is 13 g and 133 g, respectively (small amphibians are not expected to eat vertebrate prey). The default size classes for snakes are small (2 g), medium (20 g), and large (800 g). The default vertebrate prey size that medium and large snakes can consume is 25 g and 1286 g, respectively (small snakes are not expected to eat vertebrate prey). EECs are calculated for the most sensitive dietary item and size class for amphibians and snakes. For both amphibians and reptiles, the most sensitive EECs and RQs are for a 20-gram animal that consumes small herbivore mammals. If dietary RQs are more sensitive than acute dose based RQs for acute exposures, they are shown as well. Dietary-based EECs and RQs are used to characterize risk from chronic exposure. The percentages of the EECs for the different dietary items are discussed in the discussion on uncertainties (Section 6.1.1.b).

T-REX also includes the capability to calculate LD_{50}/ft^2 risk values and specialized risk analyses for granular applications of PCNB. Conceptually, an LD_{50}/ft^2 is the amount of a pesticide estimated to kill 50% of exposed animals in each square foot of applied area. Although a square foot does not have defined ecological relevance, and any unit area could be used, risk presumably increases as the number of $LD_{50}s/ft^2$ increases. The LD_{50}/ft^2 is used to estimate risk for granular applications. The LD_{50}/ft^2 is calculated using a toxicity value (adjusted LD_{50}) and the EEC (mg a.i./ft^2) and is directly compared with the Agency's LOCs. LD_{50}/ft^2 risk values were calculated for the uses summarized in **Table 3-6**. These uses represent the maximum application rates for each application type. In the case of cole crops and cotton, an 85% incorporation of granules is assumed because the label specifies that applied product should be covered with soil. Conversely, application to turf by ground spreader is not incorporated.

Table 3-6. Input Parameters for Applications Used to Derive LD_{50}/ft^2 Risk Values for PCNB with T- REX.

Use	Application Type	Application Media	Application Rate (lbs a.i./A)	% Incorporation
Cole crops	Broadcast;	Granular	22.5	85

Use	Application Type	Application Media	Application Rate (lbs a.i./A)	% Incorporation
	banded			
Cotton	In-furrow	Granular	2	85
Turf	Ground spreader	Granular	43.56	0

3.3.2. Dietary Exposure to Mammals, Birds, and Amphibians Derived Using T-REX

Upper-bound Kenaga nomogram values reported by T-REX are used for derivation of dietary EECs for the CTS (all DPS), CCR, SFGS and their potential prey (**Table 3-7**).

EECs in T-REX that are applicable to direct effects to the CCR are for small (20-g juveniles) and medium (100-g adult) birds consuming a variety of dietary items. The most sensitive EEC for the CCR is for the small bird consuming short grass. EECs in T-REX that are applicable to assess direct effect to the terrestrial-phase CTS and SFGS are for small birds (20g) consuming short grass[6]. For birds (surrogates for terrestrial-phase amphibians and reptiles), EECs and RQs for acute dose-based and chronic dietary-based exposure are calculated as these are the most sensitive values. If the LC_{50} is lower than the LD_{50}, the highest acute dietary EEC and RQ are shown as well. For mammals, EECs and RQs for acute dose based and chronic dose based exposure are calculated as these are typically the most sensitive values. If the dietary assessment results in higher RQs than the dose-based assessment, the highest dietary RQs are shown as well.

Table 3-7. Upper-bound Kenaga Nomogram EECs for Dietary- and Dose-based Exposures of Birds and Mammals Derived Using T-REX for PCNB.

Use(s), Type of Application	App Rate (lb a.i./A, # Apps, Interval (days)	EECs for Birds as Direct Effects to CCR, CTS (all DPS), SFGS and Indirect Effects to CCR, SFGS, and CTS (small birds [20 g] consuming short grass)		EECs for Mammals as Indirect Effects (Prey) for CCR, SFGS, and CTS (small mammals [15 g] consuming short grass)	
		Dietary-based EEC mg/kg-diet	Dose-based EEC mg/kg-bw	Dietary-based EEC mg/kg-diet	Dose-based EEC mg/kg-bw
Cole crops (ground spray)	22.5, 1, N/A	5400	6150	5400	5148
Cotton (ground spray)	2, 1, N/A	480	547	480	458
Potatoes	5, 1, N/A	1200	1367	1200	1144
Turf (ground spray)	33, 2, 28	12469	14200	12469	18888

N/A = not applicable; App = Application

[6] The short grass EECs and RQs are used for reptiles and amphibians to represent a conservative screen. It is not being assumed that amphibians and snakes eat short grass, the result of modeling the 20 gram bird consuming short grass is more conservative than modeling an alternative diet for amphibians and snakes and is therefore, a valid conservative screen and is protective of these species. If the short grass assessment does not result in LOC exceedances, there is a high confidence that effects are unlikely to occur.

3.3.3. Dietary Exposure to Amphibians and Reptiles Derived Using T-HERPS

Birds were used as surrogate species for terrestrial-phase CTS and SFGS. Terrestrial-phase amphibians and reptiles are poikilotherms indicating that their body temperature varies with environmental temperature. Birds are homeotherms indicating that their temperature is regulated, constant, and largely independent of environmental temperatures. As a consequence, the caloric requirements of terrestrial-phase amphibians and reptiles are markedly lower than for birds. Therefore, on a daily dietary intake basis, birds consume more food than terrestrial-phase amphibians. This can be seen when comparing the caloric requirements for free living iguanid lizards (used in this case as a surrogate for terrestrial phase amphibians) to song birds (USEPA, 1993):

$$\text{iguanid FMR (kcal/day)} = 0.0535\ (\text{bw g})^{0.799}$$

$$\text{passerine FMR (kcal/day)} = 2.123\ (\text{bw g})^{0.749}$$

With similar values of the exponents in the allometric functions, given comparable body weight, the free-living metabolic rate (FMR) of birds is roughly 40 times greater than that of reptiles, though the differences tend to narrow as body weights increase.

Because the existing risk assessment process is driven by the dietary route of exposure, a finding of safety for birds, with their much higher feeding rates and, therefore, higher potential dietary exposure is reasoned to be protective of terrestrial-phase amphibians consuming similar dietary items. For this not to be the case, terrestrial-phase amphibians would have to be approximately 40 times more sensitive than birds for the differences in dietary uptake to be negated. However, existing dietary toxicity studies in terrestrial-phase amphibians for PCNB are lacking. To quantify potential differences in food intake between birds and terrestrial-phase CTS and other amphibians, food intake equations for the iguanid lizard were used to replace the food intake equation in T-REX for birds, and additional food items of the CTS and amphibians were evaluated. These functions were encompassed in a model called T-HERPS[7]. EECs calculated using T-HERPS are shown in this section and potential risk is further discussed in the risk characterization section.

EECs in T-HERPS that are applicable to the CTS are small (2-g juveniles) amphibians consuming small and large insects and medium (20 g) amphibians consuming small and large insects, small herbivorous and insectivorous mammals, and amphibians. The dietary item that results in the highest EEC for CTS (all DPS) is the small herbivorous mammal. EECs calculated using T-HERPS for the CTS are shown in **Table 3-8**.

[7] T-HERPS is available at: http://www.epa.gov/oppefed1/models/terrestrial/index.htm

Table 3-8. Upper-bound Kenaga Nomogram EECs for Dietary- and Dose-based Exposures of Amphibians Derived Using T-HERPS for PCNB.

Use(s), Type of Application	App Rate (lb a.i./A, # App, Interval (days)	EEC for Amphibians for Direct Effects to CTS and indirect effects to SFGS (medium amphibians [20 g] consuming small herbivorous mammals [13 g])	
		Dietary-based EEC (mg/kg-diet)	Dose-based EEC (mg/kg-bw)
Cole crops (ground spray)	22.5, 1, N/A	5480	3562
Cotton (ground spray)	2, 1, N/A	487	317
Potatoes	5, 1, N/A	1218	792
Turf (ground spray)	33, 2, 28	12653	8225

N/A = not applicable; App = Application

T-REX may underestimate exposure to snakes when birds are used as a surrogate and are assumed to eat similar dietary items because of the larger meal size a snake may consume in a single day.[8] That is why birds consuming short grass in T-REX are used as the screen to determine whether further refinement in T-HERPS is needed for snakes. T-HERPS was modified (version 1.1) to estimate exposure to snakes based on the maximum sized prey item that they could consume and is used to refine risk estimates when LOCs are exceeded for small birds consuming short grass based on RQs estimated in T-REX. The following allometric equation developed by King 2002 was used to estimate the maximum size of prey items for snakes (King, 2002).

$$\text{Prey Size} = \text{Snake Mass}^{1.015}$$

The 95% confidence limits on the coefficient (exponent) are 0.959 and 1.071 (King, 2002). The upper limit was used in T-HERPS to estimate exposure to snakes.

EECs in T-HERPS that are applicable to the SFGS are small (2-g juveniles) snakes consuming small and large insects and medium-sized (20 g) snakes consuming small and large insects, small herbivorous and insectivorous mammals, and amphibians. The most sensitive EECs and RQs for SFGS are for the medium animal consuming small herbivorous mammals. EECs calculated using T-HERPS for the SFGS are shown in **Table 3-9**.

Table 3-9. Upper-bound Kenaga Nomogram EECs for Dietary- and Dose-based Exposures of Reptiles Derived Using T-HERPS for PCNB.

Use(s), Type of Application	App Rate (lb a.i./A, # App, Interval (days)	EEC for Small SFGS (small snake [2 g] consuming small insects)		EEC for Medium SFGS (medium snake [20 g] consuming herbivorous mammals [25 g])	
		Dietary-based EEC (mg/kg-diet)	Dose-based EEC (mg/kg-bw)	Dietary-based EEC (mg/kg-diet)	Dose-based EEC (mg/kg-bw)
Cole crops (ground spray)	22.5, 1, N/A	3038	169	4121	5151

[8] When examining the same application rates and types, RQs calculated in T-REX for small birds consuming short grass are higher than or equal to the highest RQs estimated in T-HERPs for medium-sized snakes consuming small herbivorous mammals. Therefore, RQs calculated in T-REX for small birds consuming short grass may be used as a conservative screen for examining risk to snakes.

Cotton (ground spray)	2, 1, N/A	270	15	366	458
Potatoes	5, 1, N/A	675	37	916	1145
Turf (ground spray)	33, 2, 28	7014	390	9515	11893

N/A = not applicable; App = Application

3.3.4. Exposure to Terrestrial Invertebrates Derived Using T-REX

T-REX is also used to calculate EECs for terrestrial invertebrates exposed to PCNB. Available acute contact toxicity data for honey bees (*Apis melifera*) exposed to PCNB (in units of µg a.i./bee), are converted to µg a.i./g (of bee) by multiplying by 1 bee/0.128 g. Dietary-based EECs calculated by T-REX for arthropods (units of µg a.i./g) are used to estimate exposure to terrestrial invertebrates (**Table 3-10**). The EECs are later compared to the adjusted acute contact toxicity data for bees in order to derive RQs.

Information on risk to arthropods is applicable to the BCB and in estimating indirect effects based on reduction in prey to the CCR, SFGS, and CTS. An example output from T-REX v. 1.4.1 is available in Appendix E.

Table 3-10. Summary EECs Used for Estimating Risk to Terrestrial Invertebrates and Derived Using T-REX for PCNB (Liquid Formulations).

Use, Method of Application	Application Rate (lbs a.i./acre), # of app, App interval (days)	Arthropod EEC (µg a.i./g)
Cole crops (ground spray)	22.5, 1, N/A	2115
Cotton (ground spray)	2, 1, N/A	188
Potatoes	5, 1, N/A	470
Turf (ground spray)	33, 2, 28	4884

N/A = not applicable; App = Application
* EEC values when less conservative 1-day foliar dissipation rate (as an alternative to default 35-day dissipation rate) is used as input in T-REX

3.3.5. Terrestrial Organisms Exposure to Residues in Aquatic Food Items

The environmental fate and ecological risk assessment written in support of the RED for PCNB provided evidence that PCNB and its degradates have potential to bioaccumulate in aquatic organisms and therefore have the potential to pose risk to mammals and birds that consume aquatic organisms. An additional assessment in 2010 used KABAM (version 1.0; USEPA, 2009) to estimate bioaccumulation of PCNB in aquatic ecosystems and subsequent exposure and risk to mammals and birds consuming aquatic organisms contaminated with the pesticide.

KABAM was used to evaluate the potential exposure and risk of direct effects to the SFGS and CCR and indirect effects to the SFGS via bioaccumulation and biomagnification in aquatic food webs. KABAM is used to estimate potential bioaccumulation of hydrophobic organic pesticides in freshwater aquatic ecosystems and risks to mammals and birds consuming aquatic organisms which have bioaccumulated these pesticides. The bioaccumulation portion of KABAM is based upon work by Arnot and Gobas (2004) who parameterized a bioaccumulation model based on PCBs and certain pesticides (*e.g.*, lindane, DDT) in freshwater aquatic ecosystems (Arnot and Gobas, 2004). KABAM relies on a chemical's octanol-water partition coefficient (K_{OW}) to estimate uptake and elimination constants through respiration and diet of organisms in different trophic levels. Pesticide tissue residues are calculated for organisms at different trophic levels in aquatic food webs. The model then uses pesticide tissue concentrations in aquatic animals to estimate dose- and dietary-based exposures and associated risks to mammals and birds (surrogate for amphibians and reptiles) consuming aquatic organisms. Seven different trophic levels including phytoplankton, zooplankton, benthic invertebrates, filter feeders, small-sized (juvenile) forage fish, medium-sized forage fish, and larger piscivorous fish, are used to represent an aquatic food web.

Aquatic tissue residues were estimated using default parameters that are considered to be representative of ecological parameters that result in conservative estimates of bioaccumulation. Specific input parameters for the CCR and SFGS on diet and body weight were also used. For the CCR, the lower range body weight estimate of 0.25 kg for this species (range 0.25 to 0.35 kg) was used based on Dunning (1984)[9]. Several reports suggest that CCR feeds on a wide variety of invertebrates, but may also scavenge dead fish (Attachment II). Since the proportion of the CCR diet consisting of either fish or benthic invertebrates is uncertain, a bounding exercise was carried out in which CCR was assumed to eat either 100% small fish or 100% benthic invertebrates. Body weight inputs for the SFGS were at the low (2 g) and high (200 g) ends of the SFGS weight inputs for T-REX. Amphibians serve as the chief dietary item for juvenile and adult SFGS; however, since KABAM does not include amphibians as a dietary item, 100% of the diet for both weight classes of SFGS was assumed to be small fish. Since the SFGS may also consume small mammals, potential indirect effects due to bioaccumulation in mammals which feed on aquatic organisms are considered. Based on a review of available information, the fog and water shrews (*Sorex sonomae* and *S. palustris*) are the only small mammals expected to feed on aquatic organisms in California. Since these species are standard in KABAM, default diets and weights were used.

Four pesticide-specific inputs are required to estimate PCNB residue concentrations in tissues of aquatic organisms: 1) log K_{OW}, 2) K_{OC}, 3) aqueous concentration of PCNB, and 4) sediment pore water concentrations of PCNB. The KABAM input values and their references are provided in **Table 3-11**. Since KABAM assumes that aquatic organisms are at steady state with respect to the chemical, it is necessary to enter surface water and benthic EECs that are representative of steady state concentrations of the chemical being modeled. The estimated time to steady state for PCNB (based on the Log K_{ow} of 5.0) is 30 days. This is supported by the observation that steady state was likely achieved within 28 days during laboratory BCF studies with fish (MRIDs 41200001 and 41951701). The selected 1-in-10 year return frequency averaging period used to represent EECs of the pesticide in sediment pore and surface waters was 21 days, which is the closest averaging period available as standard PE5 output for reporting PRZM & EXAMS model results.

[9] Dunning, J.B. 1984. Body weights of 686 species of North American Birds. Western Bird Banding Association. Monograph number 1. May 1984.

Table 3-11. Bioaccumulation Model Input Values for PCNB.

Parameter	Input Value	
Pesticide Name	PCNB	
Log K_{OW}	5.0	
K_{OC} (estimated)	6470 L/kg_{OC}	
Use patterns	**Concentration in sediment pore water (ppb)**	**Total pesticide concentration in water (ppb)**
Cole Crops	12.1	6.0
Cotton	1.1	0.39
Potatoes	1.3	0.44
Turf (foliar)	7.1	4.4
Turf (granular)	7.5	4.3

Before estimating concentrations of PCNB in different animal tissues, BCFs were estimated for different trophic levels in KABAM. The water parameter input in KABAM for dissolved oxygen concentration in water was 5 mg/L, and the water temperature was 15°C. The body characteristics of aquatic organisms (*i.e.*, wet weight, lipid content, water content) are based on the default values in KABAM[10]. KABAM-generated bioconcentration factors (BCFs) for various aquatic trophic levels are depicted in **Table 3-12**. These BCFs suggest that PCNB and its degradates can bioconcentrate in aquatic organisms.

Table 3-12. KABAM-estimated PCNB residue BCF values for aquatic organisms based on total weight.

Trophic Level	BCF (µg/kg-ww)/(µg/L)
Phytoplankton	4801
Zooplankton	3421
Benthic Invertebrates	3642
Filter Feeders	2394
Small Fish	4685
Medium Fish	4685
Large Fish	4806

Based on the bioaccumulation model, estimated concentrations of PCNB in tissues of organisms at different trophic levels, following application of the chemical to cole crops, cotton, potatoes, and turf, range from 3 to 113 mg/kg, with tissue concentrations increasing from lower to higher trophic levels (**Table 3-13**).

As discussed in Section 3.2.4.e, monitoring data for fish report PCNB concentrations ranging from 0.766 to 5.54 µg/kg, which is lower than predicted by KABAM. However, as previously discussed, these monitoring data likely underestimate PCNB's residues of concern (*i.e.*, presence of PCNB degradates such as PCTA) and were not targeted to coincide with either PCNB application areas (spatially) or application periods (temporally). Therefore, the residues measured in the fish samples likely underestimate residues in fish collected in close proximity to PCNB use sites.

[10] See KABAM User's Guide: http://www.epa.gov/oppefed1/models/water/kabam/kabam_user_guide.html

Table 3-13. Estimated concentrations (mg/kg-ww) of PCNB and its degradates in aquatic organisms at different trophic levels based on application of PCNB to different crops. Values calculated using KABAM and 21-d EECs from PRZM/EXAMS.

Trophic Level	Cole Crops (mg/kg-ww)	Cotton (mg/kg-ww)	Potatoes (mg/kg-ww)	Turf-Foliar (mg/kg-ww)	Turf-Granular (mg/kg-ww)
Phytoplankton	55	5	6	32	34
Zooplankton	42	4	5	25	26
Benthic Invertebrates	47	4	5	28	29
Filter Feeders	31	3	3	18	19
Small Fish	69	6	7	41	43
Medium Fish	81	7	9	48	51
Large Fish	113	10	12	66	70

EECs were calculated for CCR and SFGS based on consumption of aquatic organisms contaminated with PCNB (**Table 3-14**). In addition, since there is the potential for SFGS to be indirectly affected by fog and/or water shrews, which may serve as prey for the SFGS, EECs for these mammal species are also provided (see Section 3.3.5 for discussion).

Table 3-14. 1-in-10 year 21-day mean EECs for SFGS, CCR, and mammals consuming aquatic organisms contaminated with PCNB.

Use Pattern	EECs for Mammals (fog/water shrews)		EECs for CCR 100% Fish Diet	EECs for CCR 100% Benthic Invertebrate Diet	EECs for Small (2 g) SFGS	EECs for Large (200 g) SFGS
	Dose based*	Dietary based†	Dietary based†	Dietary based†	Dietary based†	Dietary based†
Cole Crops	27	47	69	47	69	69
Cotton	2.5	4.2	6.2	4.2	6.2	6.2
Potatoes	2.9	5.0	7.4	5.0	7.4	7.4
Turf (foliar)	16	28	41	28	41	41
Turf (granular)	17	29	43	29	43	43

* Dose-based EEC units are in mg/kg-bw/day
† Dietary EEC units are in mg/L

3.3.6. Terrestrial Plant Exposure Assessment

Exposure of terrestrial plants to PCNB is not calculated because there are no relevant existing data on toxicity of PCNB or its degradates to terrestrial plants.

4. Effects Assessment

This assessment evaluates the potential for PCNB to directly or indirectly affect the BCB, CTS (all DPS), DS, CCR, CFWS, SFGS, and TG or modify their designated critical habitat. Assessment endpoints for the effects determination for each assessed species include direct toxic effects on survival, reproduction, and growth, as well as indirect effects, such as reduction of the prey base or modification of its habitat. In addition, potential modification of critical habitat is assessed by evaluating effects to the PCEs, which are components of the critical habitat areas that provide essential life cycle needs of each assessed species. Direct effects to the aquatic-phase CTS are based on toxicity information for aquatic-phase amphibians, while terrestrial-phase amphibian (CTS) and reptile (SFGS) effects are based on avian toxicity data, given that birds are generally used as a surrogate for terrestrial-phase amphibians and reptiles.

As described in the Agency's Overview Document (USEPA, 2004), the most sensitive endpoint for each taxon is used for risk estimation. For this assessment, evaluated taxa include freshwater fish, aquatic-phase amphibians, freshwater invertebrates, estuarine/marine fish, estuarine/marine invertebrates, aquatic plants, birds (used as a surrogate for terrestrial-phase amphibians and reptiles), mammals, terrestrial invertebrates, and terrestrial plants. Acute (short-term) and chronic (long-term) toxicity information is characterized based on registrant-submitted studies and a comprehensive review of the open literature on PCNB and its degradates PCA and PCB.

4.1. Ecotoxicity Study Data Sources

Toxicity endpoints are established based on data generated from guideline studies submitted by the registrant, and from open literature studies that meet the criteria for inclusion into the ECOTOX database maintained by EPA/Office of Research and Development (ORD) (USEPA, 2004). Open literature data presented in this assessment were obtained from previous PCNB assessments as well as ECOTOX information obtained in November, 2011. In addition, open-literature studies for transformation products PCA and PCB were also included in ECOTOX data collection. In order to be included in the ECOTOX database, papers must meet the following minimum criteria:

(1) the toxic effects are related to single chemical exposure;
(2) the toxic effects are on an aquatic or terrestrial plant or animal species;
(3) there is a biological effect on live, whole organisms;
(4) a concurrent environmental chemical concentration/dose or application rate is reported; and
(5) there is an explicit duration of exposure.

Open literature toxicity data for other 'target' insect species (not including bees, butterflies, beetles, and non-insect invertebrates including soil arthropods and worms), which include efficacy studies, are not currently considered in deriving the most sensitive endpoint for terrestrial insects. Efficacy studies do not typically provide endpoint values that are useful for risk assessment (*e.g*., NOAEC, EC_{50}, *etc.*), but rather are intended to identify a dose that maximizes a particular effect (*e.g*., EC_{100}). Therefore, efficacy data and non-efficacy toxicological target insect data are not included in the ECOTOX open literature summary table provided in Appendix H. For the purposes of this assessment, 'target' insect species are defined as all terrestrial insects with the exception of bees, butterflies, beetles, and non-insect invertebrates (*i.e*., soil arthropods, worms, *etc*.) which are included in the ECOTOX data presented in Appendix H. The list of citations including toxicological and/or efficacy data on target insect species not considered in this assessment is provided in Appendix G.

Data that pass the ECOTOX screen are evaluated along with the registrant-submitted data, and may be incorporated qualitatively or quantitatively into this endangered species assessment, as specified by the Agency's guidance on the evaluation of open literature (USEPA, 2011). In general, effects data in the open literature that are more conservative (*i.e.*, show greater sensitivity) than registrant-submitted data are considered. The degree to which open literature data are quantitatively or qualitatively characterized for an effects determination is dependent on whether the information is relevant to the assessment endpoints (*i.e.*, survival, reproduction, and growth) identified in Section 2.8. For example, endpoints such as behavior modifications are likely to be qualitatively evaluated, because quantitative relationships between modifications and reduction in species survival, reproduction, and/or growth are not available. Although the effects determination relies on endpoints that are relevant to the assessment endpoints of survival, growth, or reproduction, it is important to note that the full suite of sublethal endpoints potentially available in the effects literature (regardless of their significance to the assessment endpoints) are considered, as they are relevant to the understanding of the area with potential effects, as defined for the action area.

Citations of all open literature that were not considered as part of this assessment because they were either rejected by the ECOTOX screen or accepted by ECOTOX but not used (*e.g*., the endpoint is less sensitive) are included in G. Appendix H also includes a rationale for rejection of those studies that did not pass the ECOTOX screen and those that were not evaluated as part of this endangered species risk assessment.

A detailed spreadsheet of the available ECOTOX open literature data, including the full suite of lethal and sublethal endpoints is presented in Appendix H. Appendix I includes a summary of the human health effects data for PCNB.

In addition to registrant-submitted and open literature toxicity information, other sources of information, including use of the acute probit dose response relationship to establish the probability of an individual effect and reviews of ecological incident data, are considered to further refine the characterization of potential ecological effects associated with exposure to PCNB. A summary of the available aquatic and terrestrial ecotoxicity information and the incident information for PCNB are provided in Sections 4.2 and 4.3.

Available data on toxicity of degradates and other stressors of concern are summarized for each taxa in the appropriate sections for the taxa. A detailed summary of the available ecotoxicity information for all PCNB degradates and formulated products can be found in Appendix F.

4.2. Toxicity of PCNB to Aquatic Organisms

Table 4-1 summarizes the most sensitive aquatic toxicity endpoints for PCNB based on an evaluation of both registrant-submitted studies and the open literature. A brief summary of submitted and open literature data considered relevant to this ecological risk assessment is presented below. Additional information is provided in Appendix F. Since a TTR approach is employed in this assessment for aquatic organisms, endpoints from PCNB degradates can be used when they are more sensitive than the parent. However, no open-literature studies on degradates were deemed to be quantitatively useful. Therefore, only endpoints for PCNB are used to estimate risk to aquatic organisms. A qualitative description of the open-literature studies for PCA and PCB are provided in this section as well as in Appendix F. All endpoints are expressed in terms of the active ingredient of PCNB or degradates unless otherwise specified.

PCNB is very highly toxic to estuarine/marine invertebrates (mysid shrimp, *Americamysis bahia*; Eastern oyster, *Crassostrea virginica*), highly toxic to freshwater fish (rainbow trout, *O. mykiss*; bluegill sunfish, *L. macrochirus*) and freshwater invertebrates (*Daphnia magna*), and moderately toxic to estuarine/marine fish (*Cyprinodon variegatus*) on an acute exposure basis. Chronic toxicity data on rainbow trout (*O. mykiss*; NOAEC = 13 µg a.i./L) and fathead minnows (*Pimephales promelas*; NOAEC = 27 µg a.i./L) revealed that growth (adult, larval) was the most sensitive endpoint in fish. In freshwater invertebrates (*D. magna*), the most sensitive endpoint following chronic exposure was number of younger per adult per day (NOAEC = 18 µg a.i./L). There are no relevant chronic toxicity data for estuarine/marine fish or invertebrates.

No aquatic plant toxicity studies have been submitted for PCNB. In addition, no relevant studies on the toxicity of PCNB or its degradates to aquatic plants have been identified in the open-literature.

Table 4-1. Aquatic Toxicity Profile for PCNB

Assessment Endpoint	Acute/ Chronic	Species	Toxicity Value for Risk Assessment (mg a.i./L)	MRID/Citation*	Study Classification (Toxicity Category)
Freshwater fish (surrogate for aquatic-phase amphibians for chronic toxicity)	Acute	Bluegill Sunfish (*Lepomis macrochirus*)	96-hr LC_{50} = 0.1 (95% CI 0.092-0.140) Slope = Not Determined[†]	40618004	Acceptable (Highly Toxic)
	Chronic	Rainbow Trout (*Oncorhynchus mykiss*)	NOAEC = 0.013 LOAEC = 0.032 Based on growth (length and weight)	41663401	Acceptable
Freshwater invertebrates	Acute	*Daphnia magna*	48-hr EC_{50} = 0.77 (95% CI 0.60-0.98) Slope = 4.8	Acc. No. 114167	Acceptable (Highly Toxic)
	Chronic	*Daphnia magna*	NOAEC = 0.018 LOAEC = 0.030 Based on number of young per adult per day	41321301	Acceptable
Estuarine/ marine fish	Acute	Sheepshead Minnow (*Cyprinodon variegatus*)	96-hr LC_{50} = 1.5 (95% CI 1.2-1.8) Slope = 2.5	40832302	Supplemental (Moderately Toxic)
	Chronic	Sheepshead Minnow (*Cyprinodon variegatus*)	NOAEC = 0.061[‡]	Acute-to-chronic ratio	
Estuarine/ marine	Acute	Mysid Shrimp (*Americamysis	96-hr LC_{50} = 0.012 (95% CI: 0.009-	40832301	Acceptable (Very Highly

Assessment Endpoint	Acute/ Chronic	Species	Toxicity Value for Risk Assessment (mg a.i./L)	MRID/Citation*	Study Classification (Toxicity Category)
invertebrates		*bahia*)	0.015) Slope = 3.1		Toxic)
	Chronic	Mysid Shrimp (*Americamysis bahia*)	NOAEC = 0.00028[§]	Acute-to-chronic ratio	
Aquatic plants	Vascular	No data	No data	No data	No data
	Non-vascular	No data	No data	No data	No data

* ECOTOX references are designated with an E followed by the ECOTOX reference number.
[†] Only one concentration had a proportion of mortality between 0 and 1.
[‡] Based on acute-to-chronic ratio (ACR) for rainbow trout applied to acute toxicity value for sheepshead minnow (ACR = 0.32 mg a.i./L [rainbow trout acute LC_{50}] ÷ 0.013 mg a.i./L [rainbow trout chronic NOAEC] = 24.6) (sheepshead minnow NOAEC = 1.5 mg a.i./L [sheepshead minnow acute LC_{50}] ÷ 24.6 [rainbow trout ACR value] = 0.061 mg a.i./L)
[§] Based on acute-to-chronic ratio (ACR) for daphnids applied to acute toxicity value for mysid shrimp (ACR = 0.77 mg a.i./L [daphnid acute LC_{50}] ÷ 0.018 mg a.i./L [daphnid chronic NOAEC] = 42.8) (Mysid Shrimp NOAEC = 0.012 mg a.i./L [mysid shrimp acute LC_{50}] ÷ 42.8 [daphnid ACR value] = 0.00028 mg a.i./L)

Toxicity to fish and aquatic invertebrates is categorized using the system shown in **Table 4-2** (USEPA, 2004). Toxicity categories for aquatic plants have not been defined.

Table 4-2. Categories of Acute Toxicity for Fish and Aquatic Invertebrates

LC_{50} (mg/L)	Toxicity Category
< 0.1	Very highly toxic
> 0.1 - 1	Highly toxic
> 1 - 10	Moderately toxic
> 10 - 100	Slightly toxic
> 100	Practically nontoxic

4.2.1. Toxicity to Freshwater Fish and Aquatic-Phase Amphibians

Freshwater fish toxicity data are used to assess potential direct effects of PCNB to TG, DS, and CTS (all DPS) and potential indirect effects of PCNB to the SFGS, CCR, and CTS (all DPS) via effects on prey. In the case of direct effects to CTS, freshwater fish serve as a surrogate for aquatic-phase amphibians since relevant PCNB data do not exist for the latter taxon. A summary of acute and chronic freshwater fish data, including data from the open literature, is provided below.

4.2.1.a. Freshwater Fish: Acute Exposure (Mortality) Studies

Acute LC_{50} values for freshwater fish species exposed to PCNB ranged from 100 to 550 µg a.i./L; therefore PCNB is classified as highly toxic to freshwater fish on an acute exposure basis. In acute toxicity testing with bluegill sunfish (MRID 40849001 and 41060801), sublethal effects including loss of equilibrium, fish at the bottom of the test chamber, and quiescence were noted at exposure concentrations ranging from 43 to 350 µg a.i./L. Similar sublethal effects were also noted in rainbow trout at PCNB exposure concentrations ranging from 130 to 450 µg a.i./L (MRID 40992701). For the purposes of this assessment, the most sensitive acute toxicity value for freshwater fish and for aquatic-phase amphibians, for which fish serve as surrogates, is the bluegill sunfish 96-hr LC_{50} of 100 µg a.i./L.

4.2.1.b. Freshwater Fish: Chronic Exposure (Growth/Reproduction) Studies

During a freshwater fish early life-stage study with rainbow trout, exposure to PCNB at 32 µg a.i./L (LOAEC) significantly reduced ($p < 0.05$) larval growth in terms of both length and weight (MRID 41663401). Significant reductions in percentage (27%) of eggs hatched and fry survival were observed at 52 µg a.i./L. The NOAEC from this study is 13 µg a.i./L

An open-literature study (Metcalfe *et al.,* 2008; ECOTOX No. 110757) was evaluated in which Japanese medaka fish (*Oryzias latipes*) were exposed to multiple PCNB concentrations ranging from 1 to 6863 µg ai/L until 17 days from the fertilized egg stage. Based on data reported in the manuscript, the proportion of fish that hatched while exposed to PCNB was statistically lower than controls at concentrations ≥ 9 µg a.i./L, resulting in a study NOAEC of 1 µg a.i./L. Deformities were also observed in developing embryos. Anisophtalmia (defined in manuscript as cases where eyes become fused or where one eye is smaller than the other or completely absent) occurred at statistically significant levels relative to control fish at concentrations of ≥686 µg a.i./L. Other abnormalities including lack of development of brain, notochord, and heart occurred at concentrations of ≥46 µg ai/L. Since the sample size in this study was lower than recommended for a fish ELS test, and since recommended measurements of environmental conditions and test substance concentrations were not provided in the manuscript, there is some uncertainty in the NOAEC value reported.

For the purposes of this assessment, the most sensitive freshwater fish chronic toxicity value is the rainbow trout NOAEC of 13 µg a.i./L.

4.2.2. Toxicity to Freshwater Invertebrates

Freshwater aquatic invertebrate toxicity data are used to assess potential direct effects of PCNB and its degradates to the CFWS and indirect effects to the SFGS, CCR, CTS (all DPS), TG, DS, and CFWS via effects on prey. A summary of acute and chronic freshwater invertebrate data, including data published in the open literature, is provided below.

4.2.2.a. Freshwater Invertebrates: Acute Exposure Studies

A freshwater invertebrate acute toxicity test (Acc. no. 114167) with PCNB was submitted using *Daphnia magna*. The 48-hour LC_{50} was 770 µg a.i./L. All daphnids exhibited lethargy in the two highest concentrations tested. Additionally, several daphnids were caught at the surface of test solution in the next two lower concentrations. PCNB is categorized as highly toxic to aquatic invertebrates on an acute exposure basis. For the purposes of this assessment, the most sensitive freshwater invertebrate acute toxicity value is the 48-hr LC_{50} of 770 µg a.i./L for *D. magna*.

Several open-literature studies on freshwater aquatic invertebrates are also available for PCB.

Chironomus riparius (Roghair *et al.,* 1994; ECOTOX No. E4072) larvae were exposed to PCB at nominal concentrations ranging from 0.10 to 3.2 mg/L for 48 hours, and mortality, appearance, and behavior were observed. When PCB was delivered through use of dimethyl sulfoxide (DMSO) as a solvent, the resulting 48-hour LC_{50} value was 0.23 mg/L. When PCB was delivered using a generator column, the resulting 48-hour LC_{50} value was >0.32 mg/L. Temperature, pH, and test substance concentration showed minimal variability during the test. This study was classified as qualitative because DMSO was used as a solvent to dissolve PCB. This solvent has been known to interfere with toxicity test results.

Gammarus pseudolimnaeus (amphipods) were exposed to PCB for 96 hours under flow-through conditions, and mortality and behavioral effects were recorded (Brooke, 1987; ECOTOX No. E14339). The 96-hour LC_{50} value was 51.1 µg/L (95% CI: 39.2 to 66.6 µg/L). During the test, amphipods were placed in the same container as fathead minnows during flow-through testing, although they were separated by wire mesh; however, it cannot be precluded that physical, chemical, or behavioral cues between amphipods and fathead minnows did not affect the outcome of the test. An additional deficiency of this study is that limited information on test substance concentrations and toxicological data were provided in the manuscript. Therefore, this study was classified as qualitative.

Hyalella azteca were exposed to PCB in beakers at time-weighted average concentrations ranging from 0.029 to 4.57 µmol/L and mortality was observed (Landrum *et al.,* 2004; E75146). This study derived LC_{50} values for time periods ranging from 26.5 to 600 hours. The resulting LC_{50} values ranged from 0.5 to 2.3 µmol/L and occurred at the longest (600 hours) and shortest (26.5 hours) time intervals, respectively. The most relevant endpoints for risk assessment are those LC_{50} values occurring around 48 to 96 hours, as these data reflect the time-frame in which acute toxicity is typically measured in OCSPP guideline studies. The lowest toxicity value from this general timeframe was the 97-hour LC_{50} of 0.84 µmol/L (210.3 µg/L). The duration and method of exposure in this assessment is in line with general practices for conducting acute toxicity studies as outline in OCSPP 850 guidelines for toxicity testing on aquatic organisms (*e.g.*, 850.1000). However, this study was classified as qualitative because multiple experiments were used to derive LC_{50} data and it is not possible to determine the results or the number of sampling points from each experiment. Consequently, it is not possible to verify LC_{50} values or calculate them for individual experiments.

The above three studies on the effects of PCB to freshwater aquatic invertebrates suggest that daphnids may not be the most sensitive taxon to PCNB and/or PCB.

4.2.2.b. Freshwater Invertebrates: Chronic Exposure Studies

Two freshwater aquatic invertebrate life-cycle tests using the TGAI were submitted for PCNB using *D. magna*. The first study (MRID 40832304) resulted in a NOAEC of 19 µg a.i./L and LOAEC of 34 µg a.i./L. Reproduction was significantly reduced at higher treatment levels. A second 21-day toxicity study (MRID 41321301) performed with the water flea estimated a

NOAEC and a LOAEC of 18 µg a.i./L and 30 µg a.i./L, respectively, based on reproductive effects. The number of young per adult per day decreased significantly with increasing concentration of PCNB. All of the test animals died after prolonged exposure at the highest treatment level of 230 µg a.i./L.

For the purposes of this assessment the most sensitive freshwater invertebrate chronic toxicity value is the NOAEC of 18 µg a.i./L for *D. magna*.

4.2.3. Toxicity to Estuarine/Marine Fish

Estuarine/marine fish toxicity data are used to assess potential direct effects of PCNB to TG and DS and indirect effects to the CCR via effects on prey. A summary of acute and chronic estuarine/marine fish data, including data published in the open literature, is provided below.

4.2.3.a. Estuarine/Marine Fish: Acute Exposure Studies

PCNB is classified as moderately toxic to estuarine/marine fish on an acute exposure basis. Three estuarine/marine fish acute toxicity tests were submitted for review with the sheepshead minnow. Two of the studies (MRID 42336901 and 40832302), were classified as supplemental. For study MRID 42336901, a 96-hr LC_{50} of 1,700 µg a.i./L was calculated; however, there is uncertainty in this endpoint because undissolved test material was observed at all concentrations except the lowest (*i.e.*, >260 µg a.i./L) and the analytical samples were not filtered/centrifuged to remove precipitate. In a second study (MRID 40832302) the 96-hr LC_{50} value was 1,500 µg a.i./L (95% CI of 1,200 to 1,800 µg a.i./L). All of the test animals died after 96 hours at the highest test concentration of 7,500 mg a.i./L. At the next lower treatment level (4,300 µg a.i./l), all surviving fish (40%) showed complete loss of equilibrium. The lowest test concentration of 1,200 µg a.i./L had mortality of 25%. In this study, precipitate was observed at all concentrations tested, resulting in uncertainty in the level of exposure to the test substance. In a third study (MRID 40882903), the 96-hour LC_{50} was 7,900 µg a.i./L. Complete loss of equilibrium, erratic swimming behavior, lethargy, and swimming at the surface were all observed in this test at concentrations ranging from 1,400 to 15,000 µg a.i./L. Precipitate was also observed in all treatment groups in this study. For the purposes of this assessment, the most sensitive estuarine/marine fish acute toxicity value is the 96-hr LC_{50} value of 1,500 µg a.i./L for the sheepshead minnow.

4.2.3.b. Estuarine/Marine Fish: Chronic Exposure Studies

No registrant-submitted chronic toxicity studies or relevant open literature with estuarine/marine fish are available for PCNB. In the absence of such data, an acute-to-chronic ratio (ACR) of 24.6, derived using the acute 96-hr LC_{50} (320 µg a.i./L) and chronic NOAEC (13 µg a.i./L) for rainbow trout, was used to estimate a chronic estuarine/marine fish NOAEC of 61 µg a.i./L (1500÷24.6=61). This estimated NOAEC value will be used to evaluate the chronic toxicity of PCNB to estuarine/marine fish.

4.2.4. Toxicity to Estuarine/Marine Invertebrates

Estuarine/marine invertebrate toxicity data are used to assess potential indirect effects of PCNB to TG DS, and CCR via effects on prey. A summary of acute and chronic estuarine/marine invertebrate data, including data published in the open literature, is provided below in Sections 4.2.3.a and 4.2.3.b.

4.2.4.a. Estuarine/Marine Invertebrates: Acute Exposure Studies

In a 96-hour acute toxicity study with mysid shrimp exposed to technical PCNB, the LC_{50} was 12 µg a.i./L (MRID 40832301). Thus, this chemical is categorized as very highly toxic to estuarine/marine crustaceans on an acute exposure basis. PCNB is also very highly toxic to mollusks (LC_{50} range: 23 µg a.i./L; MRIDs 40882902). For the purposes of this assessment, the most sensitive estuarine/marine invertebrate acute toxicity value is the 96-hr LC_{50} of 12 µg a.i./L for mysid shrimp.

4.2.4.b. Estuarine/Marine Invertebrates: Chronic Exposure Studies

No registrant-submitted chronic toxicity studies or relevant open literature with estuarine/marine invertebrates are available for PCNB. In the absence of such data, an acute-to-chronic ratio (ACR) of 43, derived using the acute 48-hr EC_{50} (770 µg a.i./L) and chronic NOAEC (18 µg a.i./L) for *D. magna*, was used to estimate a chronic estuarine/marine fish NOAEC of 0.28 µg a.i./L (12÷43=0.29). This estimated NOAEC value is used to evaluate the chronic toxicity of PCNB to estuarine/marine invertebrates.

4.2.5. Toxicity to Aquatic Plants

Aquatic plant toxicity studies are used as one of the measures of effect to evaluate whether a given pesticide may affect primary production and are used to assess the potential indirect effects of PCNB to SFGS, CCR, CTS, TG, and CFWS via effects on food and/or habitat. However, no registrant-submitted chronic toxicity studies or relevant open literature with aquatic plants are available for PCNB.

4.3. Toxicity of PCNB to Terrestrial Organisms

Table 4-3 summarizes the most sensitive terrestrial toxicity endpoints for PCNB, based on an evaluation of both registrant-submitted studies and the open literature. A brief summary of submitted and open literature data considered relevant to this ecological risk assessment is presented below. Additional information is provided in Appendix F. All endpoints are expressed in terms of the active ingredient of PCNB or degradates unless otherwise specified. For terrestrial effects, only toxicity of the parent compound is considered since EFED does not typically model residues of degradates on terrestrial food items; therefore, a TTR approach is not employed in this assessment for determining direct effects to terrestrial organisms.

PCNB is practically non-toxic to bobwhite quail (*Colinus virginianus*) and rats (*Rattus norvegicus*) on an acute exposure basis and is practically non-toxic to the mallard duck (*Anas platyrhynchos*) and bobwhite quail (*Colinus virginianus*) on a subacute dietary basis. Chronic toxicity data on bobwhite quail (NOAEC = 600 mg a.i./kg-diet) and mallard ducks (NOAEC =

600 mg a.i./kg-diet) revealed that reproduction (hatchling survival) and growth (adult body weight), respectively, were the most sensitive endpoints. Chronic toxicity testing on rats (NOAEC = 24 mg a.i./kg-diet) resulted in decreased growth of both parents and offspring.

Table 4-3. Terrestrial Toxicity Profile for PCNB.

Endpoint	Acute/ Chronic	Species	Toxicity Value for Risk Assessment (Toxiciy Classification)	MRID/ Citation	Study Classification
Birds (surrogate for terrestrial-phase amphibians and reptiles)	Acute	Bobwhite quail (*Colinus virginianus*)	LD_{50} >2,250 mg a.i./kg-bw Slope: N/A	40618001	Acceptable (Practically non-toxic)
	Subacute Dietary		LC_{50} >5,620 mg a.i./kg-diet Slope: N/A	40618003	Acceptable (Practically non-toxic)
	Chronic		NOAEC = 600 mg a.i./kg-diet LOAEC = 1200 mg a.i./kg-diet (Endpoint effect: reduced 14-day hatchling survival)	41332801	Acceptable
Mammals	Acute	Rat (*Rattus norvegicus*)	LD_{50} = >5,050 mg a.i./kg-bw Slope = N/A	41443101	(Practically non-toxic)
	Chronic		2-gen reproduction study NOAEC = 24 mg a.i./kg diet (1.2 mg/kg bw/day) LOAEC = 3,000 mg/kg-diet	41918701	Acceptable
Terrestrial invertebrates	Acute Contact	Honey bee (*Apis mellifera*)	LD_{50} >100 µg a.i./bee (>781 µg/g-bw*) Slope = N/A	40506102	Acceptable
Terrestrial plants	No data available for terrestrial plants for PCNB				

* Using the average adult honey bee weight of 0.128 g.

Acute toxicity to terrestrial animals is categorized using the classification system shown in **Table 4-4** (USEPA, 2004). Toxicity categories for terrestrial plants have not been defined.

Table 4-4. Categories of Acute Toxicity for Avian and Mammalian Studies.

Oral LD_{50}	Dietary LC_{50}	Toxicity Category
< 10 mg/kg-bw	<50 mg/kg-diet	Very highly toxic
10 - 50 mg/kg-bw	50 - 500 mg/kg-diet	Highly toxic
51 - 500 mg/kg-bw	501 - 1000 mg/kg-diet	Moderately toxic
501 - 2000 mg/kg-bw	1001 - 5000 mg/kg-diet	Slightly toxic
> 2000 mg/kg-bw	>5000 mg/kg-diet	Practically non-toxic

4.3.1. Toxicity to Birds, Reptiles, and Terrestrial-Phase Amphibians

Avian toxicity data were used to assess potential direct effects of PCNB to the SFGS, CCR, and terrestrial-phase CTS (all DPS) as well as indirect effects of PCNB to the SFGS and CCR via effects on prey. As specified in the Overview Document, the Agency uses birds as surrogates for reptiles and terrestrial-phase amphibians when toxicity data for each specific taxon are not available (USEPA, 2004). The available open literature has no information on PCNB toxicity to reptiles or terrestrial-phase amphibians. A summary of acute and chronic avian data, including sublethal effects, is provided below in Sections 4.3.1.a and 4.3.1b.

4.3.1.a. Birds: Acute/Subacute Exposure (Mortality) Studies

The acute oral toxicity of PCNB to bobwhite quail was assessed over 14 days (MRID 40618001). The 14 day-acute oral LD_{50} exceeded the highest dose tested (>2,250 mg a.i./kg-bw). There was no mortality during the study. No physiological or behavioral abnormalities were observed and body weights and the rate of food consumption remained unaltered. PCNB is classified as practically non-toxic to bobwhite quail on an acute oral exposure basis.

Four avian subacute dietary studies were submitted for review (two studies with bobwhite quail and mallard ducks). In all four studies, LC_{50} values exceeded the highest concentration tested (LC_{50} range: >5,620 to >54,000 mg a.i./kg-diet). In a 9-day quail study (MRID 40652602), marked anorexia and lethargy were observed at the two highest test concentrations. A total of five birds died during the study (one at the 27,000 mg a.i./kg-diet dose and four at the 54,000 mg a.i./kg-diet dose). Gross morphological examination of the five birds revealed no abnormalities. In a 5-day study with quail (MRID 40618003), the LC_{50} exceeded the highest concentration (5,620 mg a.i./kg-diet) tested. Anorexia and significant low weight gain were observed in the 5,620 mg a.i./kg-diet treatment group. In a 9-day mallard duck study (MRID 40652603), marked anorexia and lethargy were observed at the two highest test concentrations (27,000 and 54,000 mg a.i./kg-diet). Gross morphological examination of the seven birds that died during testing revealed abnormalities in four of the birds; three had gas-filled intestines and the fourth had what was characterized as "yolk" in the abdominal cavity. The dietary LC_{50} value was reported as greater than the highest concentration (54,000 mg a.i./kg-diet) tested. In a 5-day study with mallard ducks (MRID 40618002) the LC_{50} again exceeded the highest concentration (5,620 mg a.i./kg-diet) tested. No mortalities were observed in this experiment; however, anorexia and significantly reduced weight gain were reported in the highest treatment group. Based on all of the LC_{50} values from these studies exceeding 5000 mg/kg diet, PCNB is classified as practically non-toxic to birds on a subacute dietary exposure basis. For the purposes of this assessment, the most sensitive acute oral and subacute dietary toxicity endpoints are >2,250 mg/kg bw (LD_{50}) and >5,620 mg/kg diet (LC_{50}), respectively, based on data from bobwhite quail.

4.3.1.b. Birds: Chronic Exposure (Growth, Reproduction) Studies

Six avian reproduction studies involving dietary exposure were submitted for review. The NOAEC values for these studies range from 600 to 5,500 mg a.i./kg-diet.

In a study with bobwhite quail (MRID 43980301), the NOAEC was determined to be 1000 mg a.i./kg-diet based upon a reduction in 14-day survivor weight and a reduction in the percentage of 14-day hatchling survivors; the LOAEC was 2,500 mg a.i./kg-diet. In a study with mallard ducks (MRID 43903301), there were no statistically significant effects on any reproductive parameters. The NOAEC was determined to be 5,500 mg a.i./kg-diet (the highest concentration tested) and no LOAEC was determined.

Two additional studies with bobwhite quail (MRID 41321101) and mallard ducks (MRID 41321201) were submitted. The quail study detected significant reductions in the number of 14-day survivors, the number of eggs set, the percentage of hatchlings of eggs set and the body weight of offspring at all treatment levels. There was a significant reduction in the number of eggs laid, the number of hatchlings, viable embryos and number of eggs set at the 11,000 and 22,000 mg a.i./kg-diet treatment concentrations. Additionally, the number of viable embryos and live 15- to 20-day embryos were decreased at a dietary concentration of 22,000 mg a.i./kg-diet. This study was classified as supplemental due to an unusually high percentage of defective eggs in the control group (40.8%) and the lack of 14-day-old survivors/pen data.

The mallard duck study detected decreases in egg shell thickness and female body weight at all treatment levels. The number of eggs laid, viable embryos, live 15- to 20-day embryos and hatchlings were reduced at the 11,000 and 22,000 mg a.i./kg-diet treatment levels. The number of defective (cracked) eggs was significantly increased and the number of eggs set was decreased at the 22,000 mg a.i./kg-diet treatment concentration. The NOAECs in these studies were therefore less than the lowest concentration (<5,500 mg a.i./kg-diet) tested. The mallard duck study was also classified as supplemental due to the lack of 14-day-old survivors/pen data.

Two other avian reproduction studies with bobwhite quail (MRID 41332801) and mallard ducks (MRID 41332802) were submitted. PCNB exposure did not result in treatment-related effects on mortality, behavior, food consumption or adult body weight of bobwhite quail. Based on a reduction of 14-day survivors as a percentage of hatchlings, the only reproductive endpoint significantly affected, the NOAEC was determined to be 600 mg a.i./kg-diet, while the LOAEC was 1200 mg a.i./kg-diet. In the mallard duck stuudy, PCNB exposure did not result in treatment-related effects on reproduction, survival, behavior, or food consumption. Due to a decrease in adult body weight, the NOAEC was determined to be 600 mg a.i./kg-diet, while the LOAEC was 1200 mg a.i./kg-diet. For the purposes of this assessment, the most sensitive chronic toxicity values for birds, and for terrestrial-phase amphibians and reptiles for which birds serve as surrogates, is a NOAEC of 600 mg a.i./kg diet based on reduced hatchling survival in bobwhite quail.

4.3.2. Toxicity to Mammals

Mammalian toxicity data are used to assess potential indirect effects of PCNB to the SFGS, CCR, and CTS (all DPS) via effects on prey and to the SFGS and CTS (all DPS) via effects on habitat. A summary of acute and chronic mammalian data, including data published in the open literature, is provided below in Sections 4.3.2.a and 4.3.2.b. A more complete analysis of toxicity data to mammals is available in Appendix I, which is a copy of the Health Effects Division (HED) chapter prepared in support of the re-registration eligibility decision (RED) finalized in 2006.

4.3.2.a. Mammals: Acute Exposure (Mortality) Studies

In a registrant-submitted acute oral toxicity study (MRID 41443101), rats exposed to PCNB developed toxic symptoms that included slight piloerection, diarrhea and slight to moderate constriction of pupils at the higher treatment concentrations; all animals were asymptomatic after three days. The acute oral LD_{50} exceeded the maximum dose (>5,050 mg/kg-bw) tested; therefore, PCNB is classified as practically non-toxic to mammals on an acute oral exposure basis. For the purposes of this assessment, the most sensitive mammalian acute oral toxicity endpoint is an LD_{50}>5,050 mg a.i./kg bw.

4.3.2.b. Mammals: Chronic Exposure (Growth, Reproduction) Studies

As reported in the ecological risk assessment written in support of the RED, a 2-generation reproduction study with rats (MRIDs 43469301, 43469302, 43469303) reviewed by the Health Effects Division, indicated no treatment-related effects for parental body weight, food consumption or reproductive performance. In addition, there were no clinical signs of toxicity in the parental animals. In the P_1 generation, hepatocellular hypertrophy was observed at 1000 mg/kg in both sexes and thyroid follicular cell hypertrophy/hyperplasia was observed at 100 and 1000 mg/kg in males and at 1000 mg/kg in females. No treatment-related effects were observed in litter size, pup viability, pup body weight or pup macroscopic examinations. The LOAEL was 100 mg/kg/day in males and 1000 mg/kg/day in females based on increases in hepatocellular hypertrophy and thyroid follicular cell hypertrophy/hyperplasia. The NOAEL was 10 mg/kg/day (200 mg/kg-diet) in males and 100 mg/kg/day in females.

Additional chronic toxicity data are available for laboratory rats (*Rattus norvegicus*) exposed to PCNB. In a 2-generation study with rats (MRID 41918701), decreased body weight was observed in parents and in offspring exposed to levels as low as 169 mg/kg-bw/day (LOAEL). Decreased body weight gain was also observed at the LOAEL in parents. The resulting NOAEL for this study was 1.2 mg/kg-bw/day (equivalent to a NOAEC of 24 mg/kg-diet/day using standard FDA lab rat conversion). These data are used as a surrogate for non-target mammals. Therefore, it is assumed that chronic effects observed in laboratory rats exposed to PCNB are representative of chronic effects that may be observed in all other species of mammals exposed to PCNB residues of concern. For the purposes of this assessment, the NOAEC of 24 mg/kg diet is used to evaluate the potential effects of PCNB to rats.

4.3.3. Toxicity to Terrestrial Invertebrates

A honey bee acute contact study using the TGAI was required for PCNB because its use may result in honey bee exposure. The acute contact LD_{50} was greater than the highest (>100 µg/bee) dose tested (MRID 40506102). No sublethal effects were noted in any of the control or test animals throughout the duration of the study.

In an open-literature study by Roark and Dale (1979; ECOTOX No. E53634), earthworms were exposed to PCNB mixed into soil at a concentration of 108 µg/cm^3 and observed at 10, 29, 52, 64, and 84 days of exposure. By day 29, there was 100 percent mortality in the PCNB treatment group but only 8 percent mortality in control worms. Earthworms treated with PCNB showed little feeding activity and did not reproduce during the test. This study was classified as qualitative because EFED currently does have an established method for evaluating risk to worms.

Another open-literature study investigated the toxicity of pentachlorobenzene (PCB) and other compounds to two earthworm species, *Lumbricus rubellus* and *Eisenia andrei*, exposed through soil (Van Gestel *et al.,* 1991; ECOTOX No. E40464). As a part of this study, 2-week LC_{50} toxicity values were derived for each earthworm species. The two-week LC_{50} values for *L. rubellus* were 115 (95% CI: 109-122) and 201 (95% CI: 150-270) mg/kg in natural and artificial soils, respectively. The two-week LC_{50} values for *E. andrei* were 134 (95% CI: 100-180) and 238 (95% CI: 180-320) mg/kg in natural and artificial soils, respectively. This study was rated as qualitative because EFED currently does not have a method to evaluate risk to earthworms. Therefore, this data can only be used for characterization purposes.

For the purposes of this assessment, the most sensitive terrestrial invertebrate acute toxicity endpoint is an LD_{50}>100 µg/bee for the honeybee.

4.3.4. Toxicity to Terrestrial Plants

There are currently no registrant-submitted data or relevant open-literature studies of the effects of PCNB on terrestrial plants. Terrestrial plants may serve as dietary items of CCR, BCB, CFWS and habitat components for the SFGS, CCR, BCB, CTS (all DPS), CFWS, TG, and DS. In addition, terrestrial dicot plant data are used to evaluate a number of the PCEs associated with the critical habitat impact analysis for the BCB, CTS-CC, CTS-SB, DS, and TG. The BCB has an obligate relationship with certain dicot plants.

4.4. Incident Database Review

A review of the Ecological Incident Information System (EIIS, version 2.1), the Aggregate Incident Reports (v. 1.0) database, and the Avian Monitoring Information System (AIMS) for ecological incidents involving PCNB was completed on 2 April 2012. It should be note that a lack of reported incidents does not imply that no incidents have occurred. Incidents described here are from throughout the U.S., including outside of the action area, as they are not necessarily unique to the location.

Two incidents have been reported in EIIS for PCNB. The first incident took place in 2004 and involves a registered use of the formulated product Ridomil Gold® PC. Damage to cotton plants (cotton seeds failed to germinate) was observed following an unincorporated granular application of PCNB. The second incident reported for PCNB also involved stunted growth in soybeans. However, in this case, a fungicide containing PCNB was improperly mixed with an antimicrobial (tebuconazole), which was not approved for use on soybeans.

There are a total of sixteen incidents involving freshwater fish kills that are associated with PCP with as many as 271,000 fish dead; however, the incident reports are not detailed enough to determine the source of the PCP. It is likely that PCP at a sufficient concentration to result in acute mortality would have originated from the use of this chemical as a wood preservative. These incidents underscore the uncertainty surrounding understanding of conditions that favor the formation of PCP as a degradate of PCNB, and relative contributions from other sources of this compound. It is likely that the contribution to the environmental loads of PCP from the use of PCNB is insignificant compared with that from the industrial uses of PCP (USEPA, 2006).

5. Risk Characterization

Risk characterization is the integration of the exposure and effects characterizations. Risk characterization is used to determine the potential for direct and/or indirect effects to the BCB, CTS (all DPS), DS, CCR, CFWS, SFGS, and TG for modification to their designated critical habitat from the use of PCNB in CA. The risk characterization provides an estimation (Section 5.1) and a description (Section 5.2) of the likelihood of adverse effects; articulates risk assessment assumptions, limitations, and uncertainties; and synthesizes an overall conclusion regarding the likelihood of adverse effects to the assessed species or their designated critical habitat (*i.e.*, "no effect," "likely to adversely affect," or "may affect, but not likely to adversely affect"). In the risk estimation section, risk quotients are calculated using standard EFED procedures and models. In the risk description section, additional analyses may be conducted to help characterize the potential for risk.

5.1. Risk Estimation

Risk is estimated by calculating the ratio of exposure to toxicity. This ratio is the risk quotient (RQ), which is then compared to pre-established acute and chronic levels of concern (LOCs) for each category evaluate (Appendix C). For acute exposures to listed aquatic animals and terrestrial invertebrates the LOC is 0.05. For acute exposures to listed birds (and, thus, reptiles and terrestrial-phase amphibians) and mammals, the LOC is 0.1. The LOC for chronic exposures to animals and acute exposures to plants is 1.0.

Acute and chronic risks to aquatic organisms are estimated by calculating the ratio of exposure to toxicity using 1-in-10 year EECs in **Table 3-3** based on the label-recommended PCNB usage scenarios summarized in **Table 3-1** and the appropriate aquatic toxicity endpoint from **Table 4-1**. Acute and chronic risks to terrestrial animals are estimated based on exposures resulting from applications of PCNB (**Table 3-7** through **Table 3-9**) and the appropriate toxicity endpoint from **Table 4-3**.

If definitive acute RQ values cannot be derived because toxicity tests show no mortality at the highest test concentrations or the presence of an undefined endpoint (*i.e.*, LC/LD_{50} or NOAEC values are "greater than"), RQs are not calculated. In these cases, risk is further characterized in the risk description section by comparing appropriate EECs and available toxicity data.

5.1.1. Exposures in the Aquatic Habitat

5.1.1.a. Freshwater Fish and Aquatic-phase Amphibians

Acute risk to fish and aquatic-phase amphibians is based on 1-in-10-year peak EECs in the standard pond and the lowest acute toxicity value for freshwater fish, *i.e*, the bluegill sunfish LD_{50}=100 µg a.i./L. Chronic risk is based on the 1-in-10-year 60-day EECs and the lowest chronic toxicity value for freshwater fish, *i.e.*, the rainbow trout NOAEC=13 µg a.i./L. Risk quotients for freshwater fish based are shown in **Table 5-1**.

Based on the available toxicity data, RQs exceed the acute risk to listed species LOC (0.05) for use on cole crops and turf, but not potatoes or cotton. The chronic risk is not exceeded for any currently approved uses of PCNB. Therefore, PCNB has the potential to directly affect the listed CTS, TG, and DS for cole crop and turf uses; it also has the potentially to indirectly effect the SFGS because the listed California red-legged frog (CRLF) is one of its main prey species. Since the risk to acute non-listed species LOC is not exceeded for any use, PCNB is unlikely to indirectly affect the CCR and CTS, which may rely on fish and/or aquatic-phase amphibians during at least some portion of their life-cycle.

Table 5-1. Acute and Chronic RQs for Freshwater Fish and Aquatic-Phase Amphibian Exposure to PCNB.

Use (Application Rate)	Peak EEC (µg/L)	60-day EEC (µg/L)	Acute RQ[†]	Chronic RQ[‡]
Cole crops (22.5 lbs a.i./A)	31.0	8.7	**0.31***	0.67
Cotton (2 lbs a.i./A)	3.0	0.61	0.03	0.05
Potatoes (5 lbs a.i./A)	3.5	0.74	0.04	0.06
Turf (foliar; 33 lbs a.i./A, 2 applic.)	19.5	5.0	**0.20***	0.38
Turf (granular; 43.56 lb a.i./A, 2 applic.)	18.9	5.2	**0.19***	0.40

* Exceeds acute risk to listed species LOC (RQ $\geq$0.05)
[†] Acute RQ = use-specific peak EEC/LC_{50} (100 µg/L, bluegill sunfish).
[‡] Chronic RQ = use-specific 60-day EEC/estimated NOAEC (13 µg/L, rainbow trout).

5.1.1.b. Freshwater Invertebrates

Acute risk to freshwater invertebrates is based on 1-in-10-year peak EECs in the standard pond and the lowest acute toxicity value for freshwater invertebrates, *i.e.*, the *D. magna* EC_{50} of 770 µg a.i./L. Chronic risk is based on 1-in-10-year 21-day EECs and the lowest chronic toxicity value for freshwater invertebrates, *i.e.*, the *D. magna* NOAEC of 18 µg a.i./L. Risk quotients for freshwater invertebrates are shown in **Table 5-2**.

Based on the available toxicity data, RQs do not exceed the acute risk to listed species or chronic risk LOCs for any of the currently approved uses of PCNB. Therefore, no direct effects are predicted for CFWS and no indirect effects are predicted for the SFGS, CCR, CTS, TG, DS, or CFWS.

Table 5-2. Representative Acute and Chronic RQs for Freshwater Invertebrate Exposure to PCNB.

Use (Application Rate)	Peak EEC (µg/L)	21-day EEC (µg/L)	Acute RQ[†]	Chronic RQ[‡]
Cole crops (22.5 lbs a.i./A)	31.0	12.1	0.04	0.67
Cotton (2 lbs a.i./A)	3.0	1.1	<0.01	0.06
Potatoes (5 lbs a.i./A)	3.5	1.3	<0.01	0.07
Turf (foliar; 33 lbs a.i./A, 2 applic.)	19.5	7.1	0.03	0.39
Turf (granular; 43.56 lb a.i./A, 2 applic.)	18.9	7.5	0.02	0.42

† Acute RQ = use-specific peak EEC/EC_{50} (770 µg/L, *Daphnia magna*).
‡ Chronic RQ = use-specific 21-day EEC/NOAEC (18 µg/L, *Daphnia magna*).

5.1.1.c. Estuarine/Marine Fish

Acute risk to estuarine/marine fish is based on 1-in-10 year peak EECs in the standard pond and the lowest acute toxicity value for estuarine/marine fish, *i.e.*, the sheepshead minnow LC_{50} of 1,500 µg a.i./L. Chronic risk is based on 1-in-10-year 60-day EECs and the acute-to-chronic ratio derived NOAEC of 61 µg a.i./L for the sheepshead minnow. PCNB risk quotients for estuarine/marine fish are shown in **Table 5-3**.

Based on the available toxicity data, RQs do not exceed the acute risk to listed species or chronic LOCs for any of the currently approved uses of PCNB. Therefore, no direct effects are predicted for the TG or DS and no indirect effects are predicted for the CCR based on the estuarine/marine fish ACR value.

Table 5-3. Acute RQs for Estuarine/Marine Fish Exposure to PCNB

Use (Application Rate)	Peak EEC (µg/L)	60-day EEC (µg/L)	Acute RQ[†]	Chronic RQ[‡]
Cole crops (22.5 lbs a.i./A)	31.0	8.7	0.02	0.14
Cotton (2 lbs a.i./A)	3.0	0.61	<0.01	0.01
Potatoes (5 lbs a.i./A)	3.5	0.74	<0.01	0.01
Turf (foliar; 33 lbs a.i./A, 2 applic.)	19.5	5.0	0.01	0.08
Turf (granular; 43.56 lb a.i./A, 2 applic.)	18.9	5.2	0.01	0.09

† Acute RQ = use-specific peak EEC/LC_{50} (1500 µg/L, sheepshead minnow).
‡ Chronic RQ = use-specific peak EEC/ACR derived NOAEC for sheepshead minnow of 61 µg/L.

5.1.1.d. Estuarine/Marine Invertebrates

Acute risk to estuarine/marine invertebrates is based on 1-in-10 year peak EECs in the standard pond and the lowest acute toxicity value for estuarine/marine invertebrates, *i.e.*, the mysid shrimp LC_{50} of 12 µg a.i./L . Chronic risk is based on 1-in-10-year 21-day EECs and the acute-to-chronic ratio derived NOAEC of 0.28 µg a.i./L for the mysid shrimp. PCNB risk quotients for estuarine/marine invertebrates are shown in **Table 5-4**.

Based on the available toxicity data, RQs exceed the acute risk to listed species LOC (0.05) for all currently approved uses of PCNB, but only exceed the acute risk to non-listed species RQ (0.5) for cole crop and turf uses. RQs exceed the chronic risk LOC for all uses. Therefore, all uses of PCNB have the potential to indirectly affect the CCR, TG, and DS which rely on estuarine/marine invertebrates during at least some portion of their life-cycle.

Table 5-4. Acute RQs for Estuarine/Marine Invertebrate Exposure to PCNB.

Use (Application Rate)	Peak EEC (µg/L)	21-day EEC (µg/L)	Acute RQ†	Chronic RQ‡
Cole crops (22.5 lbs a.i./A)	31.0	12.1	**2.6***	**43*****
Cotton (2 lbs a.i./A)	3.0	1.1	**0.25****	**3.9*****
Potatoes (5 lbs a.i./A)	3.5	1.3	**0.29****	**4.6*****
Turf (foliar; 33 lbs a.i./A, 2 applic.)	19.5	7.1	**1.6***	**25*****
Turf (granular; 43.56 lb a.i./A, 2 applic.)	18.9	7.5	**1.6***	**27*****

* Exceeds acute risk to non-listed species LOC (RQ ≥0.5) and acute risk to listed species LOC (RQ ≥0.05)
** Exceeds acute risk to listed species LOC (RQ ≥0.05)
***Exceeds chronic LOC (RQ ≥1)
† Acute RQ = use-specific peak EEC/LC_{50} (12 µg/L, mysid shrimp).
‡ Chronic RQ = use-specific peak EEC/ACR derived NOAEC for mysid shrimp of 0.28 µg/L.

5.1.1.e. Aquatic Plants

There are no relevant toxicity data available for either vascular or non-vascular plants for PCNB. Therefore, risk quotients cannot be derived for aquatic plants. No species in this assessment have obligate relationships with aquatic plants; however, the CFWS, TG, DS, CTS, SFGS, and CCR may all be indirectly affected by changes to aquatic plant populations which may provide habitat or food to these species. Therefore, the lack of aquatic plant toxicity data for PCNB is a source of uncertainty in this assessment.

5.1.2. Exposures in the Terrestrial Habitat

5.1.2.a. Birds (surrogate for Reptiles and Terrestrial-phase Amphibians)

Potential direct effects to terrestrial species are based on foliar and granular applications of PCNB. Potential risks to birds (including CCR) and, thus, reptiles (including SFGS) and terrestrial-phase amphibians (including CTS), are evaluated using T-REX, acute and chronic toxicity data for the most sensitive bird species for which data are available, and the most sensitive dietary item and size class for that species. Potential for indirect effects to CCR, SFGS, and CTS (all DPS) are also considered, based on risk to birds and/or amphibians in general due to a reduction in prey. RQs for indirect effects are calculated in the same manner as those for direct effects. The most sensitive EEC calculated in T-REX is for small birds consuming short grass. As a screening-level analysis, the RQ in T-REX for small birds consuming short grass serves as the most sensitive dietary item and size class for birds, reptiles, and terrestrial-phase amphibians. Further, potential effects to birds of all sizes for granular uses of PCNB are evaluated based on LD_{50}/ft^2 (inputs in **Table 3-6**).

T-HERPS is used as a refinement to RQs for reptiles and terrestrial-phase amphibians if T-REX indicates potential risk based on birds consuming short grass. Small snakes and terrestrial-phase amphibians only consume insects while medium and large snakes and amphibians consume small and large insects, mammals, and other amphibians. The most sensitive RQ for snakes and amphibians are for medium-sized animals consuming small herbivore mammals.

Since the only available avian acute toxicity endpoint for birds is non-definitive (*i.e.*, LC/LD_{50} value is "greater than"), acute RQs cannot be calculated and direct and indirect risk to birds, reptiles, and terrestrial-phase amphibians will be further characterized in the Risk Description section of this document.

Screening-level RQs based on chronic avian toxicity data result in LOC exceedances for all PCNB uses except for cotton (**Table 5-5**). Therefore, current uses of PCNB have the potential to directly affect the CCR and indirectly affect the CTS and SFGS based on affects to birds and by extension to reptiles and terrestrial-phase amphibians.

Based on the refined T-HERPS analysis (**Table 5-6**), RQs exceed the chronic risk LOC for all uses, indicating the potential for direct effects to terrestrial-phase amphibians, including the CTS (all DPS), and reptiles, including the SFGS. Since the CTS and SFGS may both consume amphibians, there is the potential for indirect effects to both species based on a reduction of prey. In addition, since SFGS may consume other reptiles, there is also the potential for indirect effects to this species based on a reduction of prey. It should be noted that even if more likely dietary items of the CTS (and other amphibians) such as small insects are considered, the chronic dietary RQs still exceed chronic LOCs.

An analysis of the affects of bioaccumulation on CCR and SFGS is provided in Section 5.1.2.c.

Table 5-5. Chronic RQs for Direct/Indirect Effects to Birds, Reptiles, and Amphibians for PCNB Derived in T-REX (Screening Level Analysis)

Use (Application Rate)	RQs for direct effects to birds (in general) and CCR, CTS (all DPS), SFGS, and indirect effects to CCR and SFGS (small bird consuming short grass)
	Chronic Dietary Based†
Cole crops (22.5 lbs a.i./A)	**9.0***
Cotton (2 lbs a.i./A)	0.80
Potatoes (5 lbs a.i./A)	**2.0***
Turf (33 lbs a.i./A, 2 applications)	**21***

* Exceeds avian chronic LOC (RQ ≥1)
† Based on dietary-based EEC and Northern bobwhite NOAEC = 600 mg/kg-diet

Table 5-6. Acute and Chronic RQs Derived Using T-HERPS for PCNB and Amphibians and Reptiles

Use (Application Rate)	RQs for direct effects to CTS (all DPS) and amphibians and indirect effects to CTS and SFGS (medium amphibians consuming herbivorous mammals)	RQs for direct effects to SFGS and reptiles and indirect effects to SFS (medium snakes consuming herbivorous mammals)*
	Chronic Dietary Based†	**Chronic Dietary Based†**
Cole crops (22.5 lbs a.i./A)	**9.1***	**6.9***
Cotton (2 lbs a.i./A)	N/A	N/A
Potatoes (5 lbs a.i./A)	**2.0***	**1.5***
Turf (33 lbs a.i./A, 2 applications)	**21***	**16***

* Exceeds chronic LOC (RQ ≥1)
† Based on dietary-based EEC and Northern bobwhite NOAEC = 600 mg/kg-diet

5.1.2.b. Mammals

Potential direct acute effects to mammals – which may serve as prey or influence habitat for SFGS, CCR, and CTS (all DPS) – cannot be estimated because the acute oral toxicity endpoint for rats is non-definitive (*i.e.*, "greater than" LD_{50} value). Direct chronic effects to mammals, which may indirectly impact the above species, are modeled using dietary EECs (inputs in **Table 3-5**). RQs for indirect effects are calculated in the same manner as those for direct effects. The most sensitive EECs calculated in T-REX are for small mammals consuming short grass.

Chronic RQs exceed the listed and non-listed species LOC for mammals for all foliar uses of PCNB (**Table 5-7**). Based on these results, PCNB does have the potential to indirectly impact the SFGS, CCR, and CTS (all DPS) for all uses, as these species rely on mammals as prey items or for habitat during at least some portion of their life-cycle.

An analysis of the affects of bioaccumulation on mammals is provided in Section 5.1.2.c.

Table 5-7. Chronic RQs for Indirect Effects to Mammals for PCNB Derived in T-REX (Screening Level Analysis)

Use (Application Rate)	RQs for Small Mammals (small mammals consuming short grass)*
	Chronic Dietary Based†
Cole crops (22.5 lbs a.i./A)	**225***
Cotton (2 lbs a.i./A)	**20***
Potatoes (5 lbs a.i./A)	**50***
Turf (33 lbs a.i./A, 2 applications)	**520***

* Exceeds chronic LOC (RQ ≥1)
† Based on dietary-based EEC and rat NOAEC = 24 mg/kg-diet
‡ EEC values when less conservative 1-day foliar dissipation rate (as an alternative to default 35-day dissipation rate) is used as input in T-REX

5.1.2.c. Risk to CCR, SFGS, and Mammals Based on Bioaccumulation

Acute RQs based on bioaccumulation could not be calculated for birds or mammals since the acute avian and mammalian toxicity endpoints are non-definitive (*i.e.* LC/LD_{50} is a “greater than” value). Therefore, acute effects to the CCR, SFGS, and mammals from feeding on PCNB contaminated prey cannot be evaluated.

Chronic RQs for CCR, SFGS, and mammals based on KABAM are presented in **Table 5-8**. The RQ for fog and water shrews exceeded the chronic risk LOC for all uses except for cotton. Since the SFGS may potentially feed on these mammalian species, there is some limited potential for indirect effects to SFGS when its prey-base is reduced due to mammals consuming aquatic organisms contaminated with PCNB. There were no chronic LOC exceedances for CCR or SFGS feeding on aquatic organisms.

Table 5-8. RQs for chronic, exposures to mammals consuming aquatic organisms which have accumulated PCNB and its degradates. Values calculated using KABAM v1.0.

Use Pattern	Chronic RQs for Mammals (fog/water shrews)		Chronic RQs for CCR (100% Fish Diet)	Chronic RQs for CCR (100% Benthic Invertebrate Diet)	Chronic RQs for Small (2 g) SFGS	Chronic RQs for Large (200 g) SFGS
	Dose based	Dietary based	Dietary based	Dietary based	Dietary based	Dietary based
Cole Crops	**11***	**1.9***	0.12	0.08	0.12	0.12
Cotton	0.98	0.18	0.01	0.01	0.01	0.01
Potatoes	**1.2***	0.21	0.01	0.01	0.01	0.01
Turf (foliar)	**6.4***	**1.1***	0.07	0.05	0.07	0.07
Turf (granular)	**6.7***	**1.2***	0.07	0.05	0.07	0.07

* Exceeds chronic LOC (RQ ≥1)

5.1.2.d. Terrestrial Invertebrates

In order to assess the risks of PCNB to terrestrial invertebrates, the honey bee is used as a surrogate for terrestrial invertebrates. EECs (µg a.i./g of bee) calculated by T-REX for arthropods (**Table 3-10**) are divided by the calculated toxicity value for terrestrial invertebrates (**Table 4-3**). However, since the LD_{50} value for PCBN on honeybees is non-definitive (*i.e.*, the LD_{50} is greater than the highest dose tested), an RQ cannot be calculated. Instead, potential for risk to BCB from available data is further discussed in the Risk Description section of this document.

5.1.2.e. Terrestrial Plants

Generally, for indirect effects, potential effects on terrestrial vegetation are assessed using RQs from terrestrial plant seedling emergence and vegetative vigor EC_{25} data as a screen. Since the BCB has an obligate relationship with specific dicot plant species, the seedling emergence and vegetative vigor NOAEC for dicots would typically be used to calculate RQs for indirect effects to these species via potential effects to dicots. However, no EC_{25} data are available for terrestrial plants for PCNB, and this is considered a critical data gap for determining indirect effects. The impact of this issue on risk assessment is discussed further in the Risk Description section.

5.1.3. Primary Constituent Elements of Designated Critical Habitat

For PCNB, the assessment endpoints for designated critical habitat PCEs involve the same endpoints as those being assessed relative to the potential for direct and indirect effects to the listed species assessed here. Therefore, the effects determinations for direct and indirect effects are used as the basis of the effects determination for potential modification to designated critical habitat.

5.1.4. Use of Probit Slope Response Relationship to Provide Information on the Endangered Species Levels of Concern

The Agency uses the probit dose-response relationship as a tool for providing additional information on the potential for acute direct effects to individual listed species and aquatic animals that may indirectly affect the listed species of concern (USEPA, 2004). As part of the risk characterization, an interpretation of acute RQs for listed species is discussed. This interpretation is presented in terms of the chance of an individual event (*i.e.*, mortality or immobilization) should exposure at the EEC actually occur for a species with sensitivity to PCNB on par with the acute toxicity endpoint selected for RQ calculation. To accomplish this interpretation, the Agency uses the slope of the dose-response relationship available from the toxicity study used to establish the acute toxicity measures of effect for each taxonomic group that is relevant to this assessment. The individual effects probability associated with the acute RQ is based on the mean estimate of the slope and an assumption of a probit dose-response relationship. In addition to a single effects probability estimate based on the mean, upper and lower estimates of the effects probability are also provided to account for variance in the slope, if available.

Individual effect probabilities are calculated based on an Excel spreadsheet tool IECV1.1 (Individual Effect Chance Model Version 1.1) developed by the U.S. EPA, OPP, Environmental Fate and Effects Division (June 22, 2004). The model allows for such calculations by entering the mean slope estimate (and the 95% confidence bounds of that estimate) as the slope parameter for the spreadsheet. In addition, the acute RQ is entered as the desired threshold. If a probit slope is not available for a particular endpoint, a default slope of 4.5 is use. Results of the IECV1.1 calculations for the individual effect probabilities for a variety of use scenarios based on PNCB toxicity data are presented in **Table 5-9**.

Table 5-9. Individual Effect Probabilities for PCNB Exposure

Exposure Scenario	Taxa	Acute RQ	Probit Slope*	Chance of Effect (1 in…)	95% CI of Slope Chance of Effect (1 in…)
Cole crops	Freshwater Fish	0.31	4.5 (default value)	9.06E+01	95% CI of slope not reported
Cotton		0.03		2.76E+11	
Potatoes		0.04		6.33E+09	
Turf (foliar)		0.20		1.21E+03	
Turf (granular)		0.19		1.71E+03	
Cole crops	Freshwater Invertebrates	0.04	4.8	1.03E+11	4.94E+03 to 1.00E+16[†]
Cotton		<0.01		1.00E+16[†]	4.77E+06 to 1.00E+16[†]
Potatoes		<0.01		1.00E+16[†]	4.77E+06 to 1.00E+16[†]
Turf (foliar)		0.03		7.48E+12	1.71E+04 to 1.00E+16[†]

Turf (granular)		0.03		5.73E+15	1.16E+05 to 1.00E+16[†]
Cole crops	Estuarine/Marine Fish	0.02	2.5	9.25E+04	1 to 1.00E+16[†]
Cotton		<0.01		3.49E+06	1 to 1.00E+16[†]
Potatoes		<0.01		3.49E+06	1 to 1.00E+16[†]
Turf (foliar)		0.01		3.49E+06	1 to 1.00E+16[†]
Turf (granular)		0.01		3.49E+06	1 to 1.00E+16[†]

* The probit slope is sourced from the study with the most sensitive toxicity endpoint used for the RQ derivation when available. If the slope cannot be obtained from the study, a default slope of 4.5 is used.
[†] 1 in 1.00E+16 is the default minimum individual effect probability used when the z-score probability is too small to calculate in Microsoft Excel.

5.2. Risk Description

The risk description synthesizes overall conclusions regarding the likelihood of adverse impacts leading to a preliminary effects determination (*i.e.*, "no effect," "may affect, but not likely to adversely affect," or "likely to adversely affect") for the assessed species and the potential for modification of their designated critical habitat based on analysis of risk quotients and a comparison to the Level of Concern (LOC). The final No Effect/May Affect determination is made after the spatial analysis is completed at the end of the risk description, Section 5.2.7. In Section 5.2.7, a discussion of any potential overlap between areas where potential usage may result in LAA effects and areas where species are expected to occur (including any designated critical habitat) is presented. If there is no overlap of the species habitat and occurrence sections with the Potential Area of LAA Effects a No Effect determination is made.

If the RQs presented in the Risk Estimation (Section 5.1) show no direct or indirect effects for the assessed species, and no modification to PCEs of the designated critical habitat, a preliminary "no effect" determination is made, based on PCNB's use within the action area. However, if LOCs for direct or indirect effect are exceeded or effects may modify the PCEs of the critical habitat, the Agency concludes a preliminary "may affect" determination for the FIFRA regulatory action regarding PCNB. Based on this risk estimation process described above the SFGS, CCR, CTS (all DPS), TG, and DS have a preliminary "may affect" determination. A summary of the risk estimation results are provided in **Table 5-10** for direct and indirect effects to the listed species assessed here and for the PCEs of their designated critical habitat.

Table 5-10. Risk Estimation Summary for PCNB– Direct and Indirect Effects

Taxa	LOC Exceedances (Yes/No)	Description of Results of Risk Estimation	Assessed Species Potentially Affected	Species Associated with a Designated Critical Habitat that May Be Modified by the Assessed Action
Freshwater Fish and Aquatic-phase Amphibians	Non-listed Species (No)	FW fish acute RQs exceed the LOCs for listed species cole crops and turf uses; chronic LOCs are not exceeded for any uses.	Indirect Effects: SFGS, CCR, CTS (all DPS)	CTS-CC, CTS-SB, TG, DS
	Listed Species (Yes)		Direct Effects: CTS (all DPS), TG, DS	

Taxa	LOC Exceedances (Yes/No)	Description of Results of Risk Estimation	Assessed Species Potentially Affected	Species Associated with a Designated Critical Habitat that May Be Modified by the Assessed Action
Freshwater Invertebrates	Non-listed Species (No)	RQs do not exceed the acute risk to listed species or chronic LOCs for any uses.	Indirect Effects: SFGS, CCR, CTS (all DPS), CFWS, TG, DS	CTS-CC, CTS-SB, TG, DS
	Listed Species (No)		Direct Effects: CFWS	
Estuarine/Marine Fish	Non-listed Species (Uncertain)	Acute RQs do not exceed the LOC for listed or non-listed species for any uses. Insufficient data to determine chronic RQs.	Indirect Effects: CCR	TG, DS
	Listed Species (Uncertain)		Direct Effects: TG, DS	
Estuarine/Marine Invertebrates	Non-listed Species (Yes)	Acute RQs exceed the non-listed species LOC for cole crop and turf uses. Insufficient data to determine chronic RQs.	Indirect Effects: CCR, TG, DS	TG, DS
Vascular Aquatic Plants	Non-listed Species (Uncertain)	Insufficient data to determine RQs	Indirect Effects: SFGS, CCR, CTS (all DPS), CFWS, TG, DS	CTS-CC, CTS-SB, TG, DS
Non-Vascular Aquatic Plants	Non-listed Species (Uncertain)			
Birds, Reptiles, and Terrestrial-Phase Amphibians	Non-listed Species (Yes)	All uses except cotton exceed chronic LOCs for birds, reptiles, and amphibians. Could not determine acute risks of concern due to non-definitive endpoint.	Indirect Effects: SFGS, CCR	CTS-CC, CTS-SB
	Listed Species (Yes)		Direct Effects: SFGS, CCR, CTS (all DPS)	
Mammals	Non-listed Species (Yes)	Chronic RQs exceed LOCs for all uses. Acute RQs exceed non-listed species LOC for cole crop and turf uses only.	Indirect Effects: CCR, SFGS, CTS (all DPS)	CTS-CC, CTS-SB
Terrestrial Invertebrates	Listed Species (Uncertain)	Non-definitive endpoint. Insufficient data to determine RQs	Indirect Effects: SFGS, CCR, CTS (all DPS) (prey items)	CTS-CC, CTS-SB, BCB
			Direct Effects: BCB	
Terrestrial Plants -Monocots	Non-listed Species (Uncertain)	Insufficient data to determine RQs	Indirect Effects: SFGS, CTS (all DPS), CCR, CFWS, BCB (obligate), TG, DS	CTS-CC, CTS-SB, TG, DS, BCB
Terrestrial Plants - Dicots	Non-listed Species (Uncertain)			

Following a preliminary "may affect" determination, additional information is considered to refine the potential for exposure at the predicted levels based on the life history characteristics (*i.e.*, habitat range, feeding preferences, *etc.*) of the assessed species. Based on the best available information, the Agency uses the refined evaluation to distinguish those actions that "may affect, but are not likely to adversely affect" from those actions that are "likely to adversely affect" the assessed species and its designated critical habitat.

The criteria used to make determinations that the effects of an action are "not likely to adversely affect" the assessed species or modify its designated critical habitat include the following:

- Significance of Effect: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where "take" occurs for even a single individual. "Take" in this context means to harass or harm, defined as the following:
  - Harm includes significant habitat modification or degradation that results in death or injury to listed species by significantly impairing behavioral patterns such as breeding, feeding, or sheltering.
  - Harass is defined as actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.
- Likelihood of the Effect Occurring: Discountable effects are those that are extremely unlikely to occur.
- Adverse Nature of Effect: Effects that are wholly beneficial without any adverse effects are not considered adverse.

A description of the risk and effects determination for each of the established assessment endpoints for the assessed species and their designated critical habitat is provided in Sections 5.2.1 to 5.2.6. The effects determination section for each listed species assessed will follow a similar pattern. Each will start with a discussion of the potential for direct effects, followed by a discussion of the potential for indirect effects. These discussions do not consider the spatial analysis. For those listed species that have designated critical habitat, the section will end with a discussion on the potential for modification to the critical habitat from the use of PCNB. Finally, in Section 5.2.7, a discussion of any potential overlap between areas of concern and the species (including any designated critical habitat) is presented. If there is no overlap of the species habitat and occurrence sections with the Potential Area of LAA Effects a No Effect determination is made (Section 5.2.8 to 5.2.13).

5.2.1. Bay Checkerspot Butterfly

5.2.1.a. Direct Effects

Acute and chronic RQ values could not be calculated for direct effects to the BCB because there is no definitive acute toxicity endpoint for the honeybee (surrogate for BCB in this assessment) and no relevant chronic terrestrial insect data from the open literature. If EECs for arthropods were compared to the highest dose tested in the acute contact honeybee toxicity study (100 µg/bee converted to 781.25 µg/g based on assumed weight of 0.128 g/bee), the ratio would range from 0.24 to 6.3 for the various PCNB uses, indicating that there is some uncertainty surrounding risk to the BCB based on lack of definitive surrogate toxicity data. In order for there not to be risks of concern to the BCB, the definitive honeybee LD_{50} would have to be ≥12,503 μg/bee to not exceed the terrestrial invertebrate listed species LOC of 0.05; this is approximately one-tenth the weight of an adult honeybee. Therefore, due to the relatively high application rates of PCNB, there is significant uncertainty in the potential for effects to the BCB based on available toxicity information.

No bee kill incidents or other incidents involving beneficial insects have been reported for PCNB in the US.

Given the uncertainty in the risk to the BCB based on available honeybee data, combined with the lack of toxicity data on the effects of PCNB on lepidopterans, the potential for direct effects to the BCB as a result of PCNB uses is presumed.

5.2.1.b. Indirect Effects

The BCB relies on terrestrial dicot plants exclusively for both food and habitat and has an obligate relationship with certain dicots. Eggs are laid on the native dwarf plantain (*Plantago erecta*), which the larvae feed upon; if this food is not sufficient for development, the larvae may move onto purple owl's clover (*Castilleja densiflora*). The adult butterflies live on nectar, feeding on a variety of plants. The BCB inhabits grasslands on serpentine soils, such as the Montara soil series. Populations now remain only in San Mateo and Santa Clara counties.

No registrant-submitted or open-literature terrestrial plant toxicity data are available for PCNB or its degradates. Since the BCB has an obligate relationship with dicot plants, the lack of terrestrial plant data represents a significant source of uncertainty. The only two incidents reported for PCNB affected terrestrial plants. Due to the lack of effects data and occurrence of incidents in terrestrial plants, the potential for indirect effects to the BCB as a result of PCNB uses is presumed.

5.2.1.c. Modification of Designated Critical Habitat

Based on the assessment of direct and indirect effects to the BCB above, it is uncertain whether modification of designated critical habitat for the BCB may occur.

5.2.2. California Tiger Salamander (All 3 DPS)

5.2.2.a. Direct Effects

Aquatic-phase

Direct effects to the aquatic-phase CTS are estimated based on acute and chronic toxicity data from freshwater fish as a surrogate due to a lack of toxicity data for aquatic-phase amphibians. The aquatic-phase includes life stages of the CTS that occur in the aquatic environment, including egg and larval stages. It also includes submerged terrestrial-phase juveniles and adults, which spend a portion of their time in water bodies that may receive runoff and spray drift containing PCNB.

Acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the CTS for those uses. In addition, individual effect probabilities for CTS based on freshwater fish data range from 1-in-9.1 to 1-in-2.8×10^{11} across PCNB uses.

Conversely, there were no exceedances for freshwater fish and aquatic-phase amphibians on a chronic exposure basis, suggesting that risk to the aquatic-phase CTS may be limited to acute exposures. There is some uncertainty in this conclusion because an open-literature study on the effects of PCNB on medaka fish (Metcalfe *et al.,* 2008; ECOTOX No. 110757) reported reproductive effects at a concentration 13 times lower than in the rainbow trout study used to calculate chronic RQs for freshwater fish in this assessment. However, the medaka study was classified as qualitatively useful for risk assessment and is only used here to indicate uncertainty in the sensitivity of the rainbow trout endpoint.

Estimated BCFs indicate that PCNB and its degradates are expected to bioconcentrate in aquatic organisms, including aquatic plants and invertebrates. Larval CTS eat algae, snails, zooplankton, small crustaceans, and aquatic larvae and invertebrates, including mosquito larvae, for about six weeks after hatching (Hurt, 2000; and USFWS, 2000, 2003a and 2005). CTS larvae switch to larger prey after this initial period, but continue to feed on aquatic insects and other aquatic invertebrates (USFWS, 2000 and 2003a). Therefore, there is the potential for CTS to be exposed to concentrations of PCNB through diet that are higher than those modeled for the aquatic environment.

There have been no reported incidents to fish or amphibians for PCNB, but there have been reported fish kills resulting from exposure to PCP, although the source of the chemical was uncertain.

Based on the weight of evidence presented here, there is a potential for direct effects to the aquatic-phase CTS as a result of PCNB uses.

Terrestrial-phase

Potential for direct effects to the terrestrial-phase CTS are assessed based on direct acute and chronic toxicity to birds as a surrogate due to a lack of toxicity data for terrestrial-phase amphibians. In this assessment, only chronic RQ values could be calculated for birds because available avian acute toxicity endpoints are non-definitive (*i.e.*, LC/LD_{50} values are greater than the highest concentration tested); in addition, the probability of an individual effect to CTS based on avian toxicity data could not be calculated due to the non-definitive endpoint value.

Based on avian reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses except for cotton. In terms of chronic risk, EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the terrestrial-phase CTS (*i.e.*, reduce RQ values below the chronic LOC of 1).

A refinement of the chronic risks posed to the terrestrial-phase CTS was performed using the T-HERPS model. The T-HERPS model refines the EEC and RQ values based on the dietary intake rate of an amphibian rather than the dietary intake rate of a bird. However, in the case of PCNB, RQs modeled in T-HERPS are similar to those in T-REX, and do not change the chronic risk conclusions; therefore, all uses of PCNB except for cotton have the potential to directly affect the CTS. It should be noted that the default refined diet in T-HERPS for the CTS (and other amphibians) is small mammals; however, even if more likely dietary items such as small insects are considered, the chronic dietary RQs still exceed chronic LOCs.

In addition to considering the default dietary item for estimating chronic risk to the CTS (*i.e.*, amphibians feeding on small herbivorous mammals), worms were also considered as a dietary item, as the CTS is known to feed on worms and there is also toxicity data for the effects of PCB on earthworms (Van Gestel *et al.,* 1991; ECOTOX No. E40464). In this study, two earthworm species (*Lumbricus rubellus*, *Eisenia andrei*) were exposed to PCB through soil resulting in two-week LC_{50} values ranging from 115 to 238 mg/kg-dry soil depending on the species and type of soil used. In order to derive concentrations of PCNB in earthworms that could be compared to toxicity data, two steps were performed: first, soil concentrations of PCNB were generated in PRZM-EXAMS; second, soil concentrations of PCNB were translated into residues on worms using a fugacity approach (methods for deriving soil and worm residues of PCNB are described in Appendix K). The resulting concentrations in worms were 580, 1,200, and 5,700 mg/kg-bw for cotton, potato, and cole crop uses, respectively (turf was not evaluated since there is no soil incorporation for this use). Both potato and cole crop residues estimated for worms are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CTS, indicating the potential for chronic risk to the CTS for these uses of PCNB based on worm dietary items.

Although acute RQs were not calculated for terrestrial-phase CTS due to the "greater than" toxicity value, it should be noted that the highest concentration tested in the available avian acute oral toxicity study for PCNB (2250 mg a.i./kg-bw) is greater than one-tenth the acute dose-based EEC (*i.e.*, would exceed the listed species LOC of 0.1) for all uses of PCNB. The dose-based EECs for small birds feeding on short grass in T-REX range from 547 to 14,200 mg a.i./kg-bw, while refined EECs for aquatic-phase amphibians in T-HERPS range from 317 to 8,225 mg a.i./kg-bw. In both cases, acute risks of concern to listed aquatic-phase amphibians are expected. In addition, if the highest concentration tested in the avian acute oral toxicity study (2250 mg a.i./kg-bw) was used as the LD_{50} value input in LD_{50}/ft^2 calculations, LD_{50}/ft^2 values for cole crops, cotton, and turf granular uses would range from <0.01 to 14 and would exceed the acute listed species LOC in at least one weight class of birds (and aquatic-phase amphibians) for all three granular uses. Therefore, the potential for acute risk to CTS cannot be precluded based on available exposure and toxicity data.

There have been no reported incidents to birds or amphibians for PCNB.

Based on the weight of evidence presented here, there is a potential for direct effects to the terrestrial-phase CTS as a result of PCNB uses.

5.2.2.b. Indirect Effects

i. Potential Loss of Prey

CTS larvae are only able to eat small crustaceans, algae, and mosquito larvae. When they are large enough, they begin to consume aquatic insects, invertebrates and tadpoles of Pacific tree frogs, California red-legged frogs, western toads, and spadefoot toads. The terrestrial-phase CTS feed on terrestrial invertebrates, insects, frogs, and worms. Indirect effects to the CTS via loss of prey species are evaluated using toxicity data and other information gathered on freshwater invertebrates, freshwater fish, terrestrial invertebrates, and small mammals.

Freshwater Invertebrates

RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-1×10^{11} to 1-in-1×10^{16} across PCNB uses. Therefore, there is low potential for indirect effects to the CTS based on risk to freshwater invertebrate prey as a result of PCNB uses. However, it should be noted that open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Freshwater Fish

There is no evidence in the literature indicating that aquatic-phase CTS consume fish. However, indirect effects to CTS through direct effects to fish (prey items) were considered in this assessment as CTS eats other aquatic vertebrates such as frogs, and fish serve as surrogates for frogs (aquatic-phase amphibians).

As detailed in the CTS direct effects section above (5.2.2.a), freshwater fish RQs exceed the acute risk to listed species LOC only, indicating that there is little potential for direct acute risks to non-listed freshwater fish and aquatic phase amphibians that may serve as prey for the CTS. However, it should be noted that there is some uncertainty surrounding the sensitivity of the species used for chronic risk estimation for freshwater fish in this assessment (see Section 5.2.2.a).

Terrestrial Invertebrates

As detailed in the BCB direct effects section (5.2.1.a), there is insufficient information to determine potential direct effects to terrestrial invertebrates based on honeybee toxicity data. However, two open-literature studies are available for earthworms that are relevant to the CTS, since the diet of this species consists, in part, of earthworms.

One of these studies was conducted with PCNB (Roark and Dale, 1979; ECOTOX No. E53634) and reported 100 percent mortality among earthworms (*Eisenia foetida*) exposed to a single PCNB soil concentration of 108,000 mg/m^3 after 29 days; in addition, individuals showed little feeding activity and did not reproduce during the test at this soil concentration. Although EFED currently does not have a formal process for evaluating risk to earthworms, it is possible for characterization purposes to compare expected soil concentrations of PCNB to the specific concentration tested in this study. Estimated PCNB soil concentrations generated by PRZM-EXAMS (see Appendix K for methods and calculations) are 5, 7, and 78 mg/kg soil for cotton, potato, and cole crop uses, respectively. This translates to 7,800, 10,780, and 117,000 mg/m^3, respectively, indicating that the level of exposure to PCNB from use on cole crops could potentially impair the survival and reproduction of earthworm prey items.

In the second earthworm study (Van Gestel *et al.,* 1991; ECOTOX No. E40464), toxicity of PCB to two earthworm species (*Lumbricus rubellus*, *Eisenia andrei*) exposed through soil resulted in two-week LC_{50} values ranging from 115 to 238 mg/kg-dry soil depending on the species and type of soil used. These toxicity values are all greater than the expected soil concentrations of PCNB from modeling results, as presented in the previous paragraph.

Based on the available information, there is some potential for indirect effects to CTS due to effects on worm prey items.

Amphibians

As described in the above section on direct effects to the CTS (Section 5.2.2.a), there is the potential for chronic risks to amphibians based on the refined analysis in T-HERPS model. Since the CTS is known to consume other amphibians, there is the potential for indirect effects to the CTS due to a reduced prey base.

ii. Potential Modification of Habitat

The CTS inhabits low elevation vernal pools and seasonal ponds and associated grassland, oak savannah, and coastal scrub plant communities. Juvenile and adult CTS spend the dry summer and fall months in the burrows of California ground squirrels (*Spermophilus beecheyi*) and Botta's pocket gopher (*Thomomys bottae*). The CTS cannot dig their own burrows, and as a result their presence is associated with active burrows of these small mammals. Indirect effects to the CTS through potential modification of habitat are evaluated based on impacts of PCNB on aquatic plants, terrestrial plants, and small mammals.

Aquatic Plants

Aquatic plants serve several important functions in aquatic ecosystems. Non-vascular aquatic plants are primary producers and provide the autochthonous energy base for aquatic ecosystems. Vascular plants provide structure, rather than energy, to the system, as attachment sites for many aquatic invertebrates, and refugia for juvenile organisms, such as fish and frogs. Emergent plants help reduce sediment loading and provide stability to nearshore areas and lower stream banks. In addition, vascular aquatic plants are important as attachment sites for egg masses of aquatic species.

Since no relevant data on aquatic plants have been identified for PCNB or its degradates, it is not possible to determine indirect effects to the CTS based on this habitat component.

Terrestrial Plants

Terrestrial plants serve several important habitat-related functions for the listed assessed species. In addition to providing habitat and cover for invertebrate and vertebrate prey items of the listed species assessed, terrestrial vegetation also provides shelter and cover from predators while foraging. Upland vegetation including grassland and woodlands provides cover during dispersal. Riparian vegetation helps to maintain the integrity of aquatic systems by providing bank and thermal stability, serving as a buffer to filter out sediment, nutrients, and contaminants before they reach the water body, and serving as an energy source.

Since no relevant data on terrestrial plants have been identified for PCNB or its degradates, it is not possible to determine indirect effects to the CTS based on this habitat component.

Small Mammals

Juvenile and adult CTS rely on the burrows of small mammals for habitat in the dry summer and fall months.

Acute mammalian RQs could not be calculated in T-REX because the available acute oral toxicity study for mammals is non-definitive (*i.e.*, LC/LD_{50} values are greater than the highest concentration tested). In addition, probability of individual effect values for mammals could also not be calculated for the same reason. Based on submitted rat two-generation reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses, indicating that small mammal prey may be adversely affected by PCNB at current use rates. In terms of chronic risk, EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammals.

Based on bioaccumulation analysis, there is high potential for mammals that consume aquatic organisms to accumulate sufficient levels of PCNB through diet to constitute a risk of concern via bioaccumulation. However, it should be noted that fog and water shrews are the only two mammal species that are known to both forage on aquatic organisms and occupy ranges that overlap with the species in this assessment.

Based on the available information, there is the potential for indirect effects to CTS habitat based on chronic risk to small mammals.

5.2.2.c. Modification of Designated Critical Habitat

Based on the assessment of indirect effects to the CTS via effects on small mammals in Section 5.2.2.b, and the lack of effects data on aquatic and terrestrial plants, there is the potential for modification of designated critical habitat for the CTS-CC and CTS-SB to occur. There is no designated critical habitat for the CTS-SC.

5.2.3. California Clapper Rail

5.2.3.a. Direct Effects

The potential for direct effects to the CCR are assessed based on direct acute and chronic toxicity effects to birds. In this assessment, only chronic RQ values could be calculated for birds because available avian acute toxicity endpoints are non-definitive (*i.e.*, LC/LD_{50} values are greater than the highest concentration tested). In addition, the probability of an individual effect to CCR based on avian toxicity data could not be calculated due to the non-definitive endpoint value.

Based on avian reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses except for cotton. In terms of chronic risk, EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the CCR (*i.e.*, reduce RQ values below the chronic LOC of 1). In addition, as described in Section 5.2.1.a, when worms are considered as a dietary item for the CCR, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CCR, indicating the potential for chronic risk to the CCR for these crop uses of PCNB based on worm dietary items.

Although acute RQs were not calculated for the CCR in T-REX due to the “greater than” toxicity value, it should be noted that the highest concentration tested in the available avian acute oral toxicity study for PCNB (2250 mg a.i./kg-bw) is greater than one-tenth the acute dose-based EEC (*i.e.*, would exceed the listed species LOC of 0.1) for all uses of PCNB. The dose-based EECs for small birds feeding on short grass range from 547 to 14,200 mg a.i./kg-bw. In addition, if the highest concentration tested in the avian acute oral toxicity study (2250 mg a.i./kg-bw) was used as the LD_{50} value input in LD_{50}/ft^2 calculations, LD_{50}/ft^2 values for cole crops, cotton, and turf granular uses would range from <0.01 to 14 and would exceed the acute listed species LOC in at least one weight class of birds for all three granular uses. Therefore, the potential for acute risks of concern to CCR cannot be excluded based on available toxicity data for birds.

During acute oral and subacute dietary toxicity tests the following sublethal effects were observed: marked anorexia, lethargy, reduced weight gain, gas-filled intestines, and yolk in the abdominal cavity.

Based on bioaccumulation analysis, there is little potential for birds to consume sufficient amounts of aquatic organisms to constitute a risk of concern for the CCR via bioaccumulation.

There have been no reported incidents to birds or amphibians for PCNB.

Based on the weight of evidence presented here, there is potential for direct effects to the CCR as a result of PCNB uses.

5.2.3.b. Indirect Effects

i. Potential Loss of Prey

The CCR are generalist and opportunistic feeders that forage at the upper end of marshes, along the ecotone between mudflat and higher vegetated zones, and in tidal sloughs. Mussels, clams, arthropods, snails, worms and small fish are its preferred foods, which it retrieves by probing and scavenging the surface while walking. The bird will only forage on mudflats or very shallow water where there is taller plant material nearby to provide protection at high tide. Although CCRs typically consume invertebrates, they have also been known to occasionally consume small birds and mammals, including the salt marsh harvest mouse. The CCR diet may contain up to 15% plant material.

Indirect effects to the CCR via loss of prey species and plant foods are evaluated using toxicity data and other information gathered on freshwater fish, freshwater invertebrates, estuarine/marine fish, estuarine/marine invertebrates, aquatic plants, birds, small mammals, terrestrial invertebrates, and terrestrial plants.

Freshwater Fish

As detailed in the CTS direct effects section above (5.2.2.a), freshwater fish RQs exceed the acute risk to listed species LOC only, indicating that direct acute risks to non-listed freshwater fish are unlikely. Therefore, there is little potential for indirect effects to the CCR based on this prey component. However, it should be noted that there is some uncertainty surrounding the sensitivity of the species used for chronic risk estimation for freshwater fish in this assessment (see Section 5.2.2.a).

Freshwater Invertebrates

Acute RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-1×10^{11} to 1-in-1×10^{16} across PCNB uses. Therefore, there is low potential for indirect effects to the CCR based on risk to freshwater invertebrate prey as a result of PCNB uses. However, it should be noted that open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Estuarine/Marine Fish

No acute or chronic RQ values representing any uses of PCNB exceed LOCs for estuarine/marine fish. The probability of an individual effect for estuarine/marine fish ranges from 1-in-9.3×10^4 to 1-in-3.5×10^6 across PCNB uses. Therefore, indirect effects are not anticipated to the CCR based on this prey component.

Estuarine/Marine Invertebrates

Acute RQ values exceed the non-listed species LOC (0.5) for estuarine/marine invertebrates for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively.

There have been no reported incidents for estuarine/marine invertebrates involving PCNB; however, incidents on invertebrates are unlikely to be noticed and reported to the Agency.

Based on the weight of evidence, there is the potential for indirect effects to the CCR based on this prey component.

Aquatic Plants

As detailed above (see Section 5.2.2.b), there are no data with which to assess toxicity of PCNB to aquatic plants. Therefore, it is not possible to determine indirect effects to the CCR based on this habitat component.

Birds

As detailed in the CCR direct effects section above, the chronic risk LOC is exceeded for all PCNB uses except for cotton. In terms of chronic risk, EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the CCR.

Based on the chronic LOC exceedances, there is the potential for indirect effects to the CCR based on risk to small avian prey.

Small Mammals

As detailed in the CTS indirect effects section above (5.2.2.b), the chronic risk LOC for mammals is exceeded for all PCNB uses based on exposure through terrestrial dietary items as well as bioaccumulation of PCNB in aquatic dietary items, indicating that small mammals may be adversely affected by PCNB at current use rates. In terms of chronic risk, EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammal prey items.

Based on the chronic LOC exceedances, there is the potential for indirect effects to the CCR based on risk to small mammal prey.

Terrestrial Invertebrates

As detailed in Section 5.2.2.b above, there is insufficient information to determine potential direct effects to terrestrial invertebrates as a result of PCNB uses based on honeybee data. However, open-literature data does suggest there is some potential for indirect effects to the CCR based on effects to worm prey items.

Terrestrial Plants

As detailed in the CTS indirect effects section above (5.2.2.b), no relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the CCR based on this prey item.

ii. Potential Modification of Habitat

The CCR inhabits cordgrass marshes around San Francisco Bay. CCR juveniles can disperse a sufficient distance to be found in both residential and agricultural areas east of SF Bay and along the open coast.

Due to the lack of effects data on aquatic and terrestrial plants, it is uncertain whether modification of designated critical habitat for the CCR may occur.

5.2.4. California Freshwater Shrimp

5.2.4.a. Direct Effects

The potential for direct effects to the CFWS are assessed based on direct acute and chronic toxicity effects to freshwater invertebrates. Acute RQs do not exceed the acute risk to non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-$1x10^{11}$ to 1-in-$1x10^{16}$ across PCNB uses. Therefore, direct effects are not expected to the CFWS based on risk to freshwater invertebrate prey under the exposure scenarios evaluated in this assessment and based on available toxicity data. It should be noted that there is significant uncertainty associated with this conclusion because of the higher toxicity of PCNB to mysid shrimp (estuarine/marine crustacean) compared to *Daphnia magna* (freshwater crustacean). Mysid shrimp are more closely related to the CFWS, and may indicate that effects to the CFWS may be underestimated based on the toxicity endpoint from *D. magna*. In addition, open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs for freshwater invertebrates in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species. Moreover, if any of the other freshwater invertebrate species were used to calculate RQs for the CFWS, it would result in exceedance of the acute risk to listed species LOC.

There have been no reported incidents for freshwater invertebrates involving PCNB; however, incidents on invertebrates are unlikely to be noticed and reported to the Agency.

Estimated BCFs indicate that PCNB and its degradates are expected to bioconcentrate in aquatic organisms, including aquatic plants and invertebrates. CFWS predominately feeds on detritus that may be colonized by algae and zooplankton within which PCNB may bioconcentrate. EFED currently does not have a model to evaluate toxicity to invertebrates or fish feeding on other organisms in aquatic systems. Therefore, lack of information on the potential for CFWS exposure to PCNB via diet represents an uncertainty.

Based on toxicity data from daphnids there is low potential for direct effects to the CFWS as a result of PCNB uses. However, there is significant uncertainty associated with this conclusion based on higher toxicity of PCNB to mysid shrimp and several freshwater invertebrates and the potential for bioaccumulation of PCNB in CFWS aquatic dietary items.

5.2.4.b. Indirect Effects

The CFWS relies on aquatic and terrestrial plants for both food and habitat. The CFWS feeds on decomposing vegetation and other detritus, consuming minute diverse particles conveyed by currents to downstream pools, which includes zooplankton. The CFWS is found only in low elevation perennial streams or intermittent streams with perennial pools in the northern San Francisco Bay Area. Freshwater shrimp require low gradient streams with diverse habitat structure including undercut banks, exposed roots, woody debris and overhanging vegetation. Indirect effects to the CFWS via loss of food and habitat are evaluated using toxicity data and other information gathered on freshwater invertebrates, aquatic plants, and terrestrial plants.

Freshwater Invertebrates

As detailed in the CTS indirect effects section (5.2.2.b), RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-$1x10^{11}$ to 1-in-$1x10^{16}$ across PCNB uses. Therefore, there is low potential for indirect effects to the CFWS based on this prey component under the exposure scenarios evaluated. However, it should be noted that open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Aquatic Plants

As detailed in the CTS indirect effects section above (5.2.2.b), no relevant data on aquatic plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the CFWS based on this prey item or habitat component.

Terrestrial Plants

As detailed in the CTS indirect effects section above (5.2.2.b), no relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the CFWS based on this prey item.

5.2.5. San Francisco Garter Snake

5.2.5.a. Direct Effects

Potential for direct effects to the SFGS are assessed based on direct acute and chronic toxicity effects to birds as a surrogate due to a lack of toxicity data for reptiles. In this assessment, only chronic RQ values could be calculated for birds because available avian acute toxicity endpoints are non-definitive (*i.e.*, LC/LD_{50} values are greater than the highest concentration tested). In addition, the probability of an individual effect to SFGS based on avian toxicity data could also not be calculated for the same reason.

Based on avian reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses except for cotton. In terms of chronic risk, EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the SFGS (*i.e.*, reduce RQ values below the chronic LOC of 1).

A refinement of the chronic risks posed to the SFGS was performed using the T-HERPS model. Avian RQ values used as screening surrogates for reptiles typically overestimate risks to reptiles. Overestimation is due to the higher energy requirements of birds over reptiles of the same body weight, which results in a higher daily food intake rate value and a resultant higher dose-based exposure for birds than would occur for a reptile of the same body weight. The T-HERPS model refines the EEC and RQ values based on dietary intake rate of a reptile, rather than a dietary intake rate of a bird. In the case of PCNB, chronic RQs show a slight decrease when modeled in T-HERPS, but remain above the listed species LOC for all use scenarios except for the lowest use (cotton).

In addition, as described in Section 5.2.1.a, when worms are considered as a dietary item for the SFGS, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the SFGS, indicating the potential for chronic risk to the SFGS for these crop uses of PCNB based on worm dietary items.

Although acute RQs were not calculated for SFGS in T-REX due to the “greater than” toxicity value, it should be noted that the highest concentration tested in the available avian acute oral toxicity study for PCNB (2250 mg a.i./kg-bw) is greater than one-tenth the acute dose-based EEC for birds (*i.e.*, would exceed the listed species LOC of 0.1) for all uses of PCNB. The dose-based EECs for small birds feeding on short grass range from 547 to 14,200 mg a.i./kg-bw, while refined EECs for snakes feeding on small herbivorous mammals range from 458 to 11,893 mg a.i./kg-bw. In both cases, acute risks of concern to reptiles are expected. In addition, if the highest concentration tested in the avian acue oral toxicity study (2250 mg a.i./kg-bw) was used as the LD_{50} value input in LD_{50}/ft^2 calculations, LD_{50}/ft^2 values for cole crops, cotton, and turf granular uses would range from <0.01 to 14 and would exceed the acute listed species LOC in at least one weight class of birds (and reptiles) for all three granular uses. Therefore, the potential for acute risk to SFGS cannot be precluded based on the available exposure and toxicity data.

There have been no reported incidents to birds or reptiles for PCNB.

Based on the weight of evidence presented here, there is a potential for direct effects to the SFGS as a result of PCNB uses.

5.2.5.b. Indirect Effects

i. Potential Loss of Prey

Adult SFGS feed primarily on California red-legged frogs and juvenile bullfrogs. Newborn and juvenile snakes prey upon Pacific tree frogs. Small mammals, reptiles, amphibians, terrestrial and aquatic invertebrates, and some fish species may also be consumed by the SFGS if they can be captured in shallow water. Indirect effects to the SFGS via loss of prey species are evaluated using toxicity data and other information gathered on freshwater fish, freshwater invertebrates, birds, small mammals, and terrestrial invertebrates.

Freshwater Fish and Aquatic-phase Amphibians

As detailed in the CTS direct effects section above (5.2.2.a), freshwater fish RQs exceed the acute risk to listed species LOC only, indicating that direct acute risks to non-listed freshwater fish is unlikely. Therefore, there is low potential for indirect effects to the SFGS based on this prey component. However, it should be noted that there is some uncertainty surrounding the sensitivity of the species used for chronic risk estimation for freshwater fish in this assessment (see Section 5.2.2.a).

Freshwater Invertebrates

RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-$1x10^{11}$ to 1-in-$1x10^{16}$ across PCNB uses. Therefore, there is low potential for indirect effects to the SFGS based on this prey component. However, it should be noted that open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Birds, Terrestrial-phase Amphibians, and Reptiles

As detailed in the CCR direct effects section above (5.2.3.a), the chronic risk LOC for birds is exceeded for all PCNB uses except for cotton. SFGS are not known to prey on birds, but avian species are used as a surrogate here for terrestrial-phase amphibians and reptiles. Moreover, refined dietary analyses for reptiles (Section 5.2.5.a) and terrestrial-phase amphibians (Section 5.2.2.a) in T-HERPS results in the same LOC exceedances as determined for birds in T-REX.

Based on the chronic LOC exceedances, there is the potential for indirect effects to the SFGS based on risk to terrestrial-phase amphibian and reptile prey species.

Small Mammals

As detailed in the CTS indirect effects section above (5.2.2.b), acute mammalian RQs could not be calculated in T-REX because the available acute oral toxicity study for mammals is non-definitive (*i.e.*, LC/LD_{50} values are greater than the highest concentration tested). In addition, probability of individual effect values for mammals could also not be calculated for the same reason.

As detailed in the CTS indirect effects section above (5.2.2.b), the chronic risk LOC for mammals is exceeded for all PCNB uses based on exposure through terrestrial dietary items as well as bioaccumulation of PCNB in aquatic dietary items, indicating that small mammals may be adversely affected by PCNB at current use rates. In terms of chronic risk, EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammal prey items.

There have been no reported incidents to mammals for PCNB.

Based on the chronic LOC exceedances, there is the potential for indirect effects to the SFGS based on risk to small mammal prey.

Terrestrial Invertebrates

As detailed in Section 5.2.2.b above, there is insufficient information to determine potential direct effects to terrestrial invertebrates as a result of PCNB uses based on honeybee data. However, open-literature data does suggest there is some potential for indirect effects to the SFGS based on effects to worm prey items.

ii. Potential Modification of Habitat

The SFGS inhabits densely vegetated ponds near open hillsides where it can sun, feed, and find cover in rodent burrows as well as forage extensively in aquatic habitats. Freshwater habitats include natural and manmade (e.g. stock) ponds, slow moving streams, vernal pools and other ephemeral or permanent water bodies which typically support inundation during winter rains. Upland habitats are within 200 ft of the mean high water mark of such aquatic habitats.

Aquatic Plants

As detailed above (see Section 5.2.2.b), there are no data with which to assess toxicity of PCNB to aquatic plants. Therefore, it is not possible to determine indirect effects to the SFGS based on this habitat component.

Terrestrial Plants

As detailed in the CTS indirect effects section above (5.2.2.b), no relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the SFGS based on this habitat component.

Small Mammals

SFGS rely on the burrows of small mammals for shelter and aestivation when ponds become dry. SFGSs may also forage for amphibians in the rodent burrows during the summer.

As detailed in the CTS indirect effects section above (5.2.2.b), the chronic risk LOC for mammals is exceeded for all PCNB uses based on exposure through terrestrial dietary items as well as bioaccumulation of PCNB in aquatic dietary items, indicating that small mammals may be adversely affected by PCNB at current use rates. In terms of chronic risk, EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammals that may affect SFGS habitat.

Based on the chronic LOC exceedances, there is the potential for indirect effects to the SFGS based on risk to small mammal-created habitat.

5.2.6. Delta Smelt and Tidewater Goby

5.2.6.a. Direct Effects

Direct effects to TG and DS are based on acute and chronic toxicity data from freshwater fish. In this assessment, acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the TG and DS. In addition, individual effect probabilities for TG and DS based on surrogate freshwater fish data range from 1-in-9.1 to 1-in-$2.8x10^{11}$ across PCNB uses.

Conversely, there were no exceedances for freshwater fish on a chronic exposure basis, suggesting that risk to the aquatic-phase TG and DS may be limited to acute exposures. There is some uncertainty in this conclusion because an open-literature study on the effects of PCNB on medaka fish (Metcalfe *et al.,* 2008; ECOTOX No. 110757) reported reproductive effects at a concentration 13 times lower than in the rainbow trout study used to calculate chronic EECs for freshwater fish in this assessment. However, the medaka study was classified as qualitatively useful for risk assessment and is only used here to indicate uncertainty in the sensitivity of the rainbow trout endpoint.

Estimated BCFs indicate that PCNB and its degradates are expected to bioconcentrate in aquatic organisms, including invertebrates, which are the primary food source for the TG and DS. Therefore, there is the potential for TG and DS to be exposed to concentrations of PCNB that are higher than those modeled for the aquatic environment, which represents an uncertainty in this assessment.

There have been no reported incidents to fish for PCNB.

Based on the weight of evidence presented here, there is the potential for direct effects to the TG and DS as a result of PCNB uses.

5.2.6.b. Indirect Effects

i. Potential Loss of Prey

As discussed in Attachment II, the diet of DS is composed primarily of zooplankton, particularly copepods. The diet of the TG consists of macroinvertebrates such as mysid shrimp, gammarids, amphipods, ostracods, and aquatic insects. Food items of the smallest tidewater gobies, which are 4-8mm (0.2-0.3 in.) in size, have not been examined, but they likely feed on unicellular phytoplankton or zooplankton like many other early stage larval fishes. Therefore, freshwater and estuarine invertebrates as well as unicellular aquatic plants are considered as prey groups for determining indirect effects to the DS and TG caused by direct effects to its prey.

Aquatic invertebrates

For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-$1x10^{11}$ to 1-in-$1x10^{16}$ across PCNB uses. Therefore, indirect effects are not expected for the TG or DS based on risk to freshwater invertebrate prey under the exposure scenarios evaluated in this assessment and based on available toxicity data. However, it should be noted that open-literature studies on freshwater benthic invertebrate species with PCB reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Conversely, for estuarine/marine invertebrates, acute RQ values exceed the non-listed species LOC (0.5) for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively.

Based on the weight of evidence, there is the potential for indirect effects to the TG and DS based on this prey component.

Non-vascular aquatic plants

As detailed above (see Section 5.2.2.b), there are no data with which to assess toxicity of PCNB to aquatic plants. Therefore, it is not possible to determine indirect effects to the TG or DS based on this prey item.

ii. Potential Modification of Habitat

Aquatic plants serve several important functions in aquatic ecosystems. Non-vascular aquatic plants are primary producers and provide the autochthonous energy base for aquatic ecosystems. Vascular plants provide structure, rather than energy, to the system, as attachment sites for many aquatic invertebrates, and refugia for juvenile organisms. Emergent plants help reduce sediment loading and provide stability to near-shore areas and lower stream banks. In addition, vascular aquatic plants are important as attachment sites for egg masses of DS.[11]

Since aquatic or terrestrial plant toxicity data have not been identified for PCNB or its degradates, it is not possible to determine the potential for modification of habitat due to indirect effects to plants. This represents a source of uncertainty in this assessment.

5.2.6.c. Modification of Designated Critical Habitat

Based on the assessment of indirect effects to the CTS via effects on estuarine/marine invertebrate prey and lack of data on potential impacts to aquatic and terrestrial plants outlined in Section 5.2.6.b, it is possible that modification of designated critical habitat for the TG and DS may occur.

5.2.7. Spatial Extent of Potential Effects

Since LOCs are exceeded for SFGS, CCR, CTS (all DPS), TG, and DS, and there are major uncertainties associated with direct and indirect effects to BCB and CFWS, an analysis of the spatial extent of potential LAA effects is needed to determine where effects may occur in relation to treated sites. If the potential area of usage and subsequent Potential Area of LAA Effects overlaps with BCB, CTS (all DPS), CCR, CFWS, SFGS, TG, or DC habitat or areas of occurrence or critical habitat of the BCB, TG, DS, CTS-CC, or CTS-SB, a likely to adversely affect determination is made. If the Potential Area of LAA Effects and the habitat and areas of occurrence or critical habitat do not overlap, a no effect determination is made.

The properties of PCNB lend itself to long-range atmospheric transport, which may lead to potential exposure of non-target organisms distant from registered use sites and resultant accumulation and magnification of residues in food chains. At this time, EFED does not have an approved model for estimating long-range atmospheric transport of pesticides and resulting exposure to organisms in areas receiving pesticide deposition from the atmosphere. The extent to which PCNB may be deposited from the air to the action area is therefore unknown. Based on this uncertainty, the entire state of California, including the San Francisco Bay region, is considered to have residues of PCNB that could potentially affect the species evaluated in this

[11] TG lay eggs in burrows (Attachment III)

assessment. As a result, RQs used to determine direct and indirect effects have not been adjusted based on proximity to pesticide application sites.

The inclusion of the entire state of California as the action in this assessment is considered conservative since exposure and associated risks to these species and their resources are expected to decrease with increasing distance away from the treated field or site of application. In addition, all of the PCNB uses evaluated in this assessment, except for the foliar use on turf, have some degree of soil incorporation specified on the label, which may decrease the amount of material that is volatilized. Also, the assumption that PCNB is persistent and subject to atmospheric transport is partly due to variability in available data and in some cases because of the lack of acceptable data (a more detailed discussion of data gaps and resulting assumptions is provided in Sections 2.10.4 and 6.2.1). Nevertheless, there is evidence from monitoring data (Section 3.2.4) that long-range transport of PCNB away from the site of application is a legitimate concern.

5.2.8. Bay Checkerspot Butterfly

In this assessment, honey bees are used as a surrogate for determining potential effects to the BCB. Although PCNB is considered practically non-toxic to honey bees, there is uncertainty associated risk conclusions based on surrogate data since the only available honey bee toxicity study did not test up to high enough doses to preclude risks of concern based on approved PCNB application rates. In addition, indirect effects from impacts on food and habitat are also uncertain due to lack of toxicity data for terrestrial plants. Therefore, the Agency makes a **may affect, and likely to adversely affect** determination and a **habitat modification determination** for the BCB, based on the uncertainty concerning direct effects and effects to the PCEs of critical habitat.

5.2.9. California Tiger Salamander (All 3 DPS)

PCNB is expected to directly impact the CTS based on toxicity to both aquatic**-** and terrestrial**-**phase amphibians, using freshwater fish and avian surrogate species data. Indirect effects from impacts on prey are also anticipated based on toxicity to mammals, freshwater fish, and possibly terrestrial invertebrates and freshwater invertebrates. Indirect effects from impacts on habitat are anticipated due to effects on mammal burrow availability. Small mammals are essential in creating the underground habitat that juvenile and adult CTS depend upon for food, shelter, and protection from the elements and predation.

Therefore, the Agency makes a **may affect, and likely to adversely affect** determination for the CTS (all DPS) and a **habitat modification determination** for the designated critical habitat of the CTS-CC, and CTS-SB based on the potential for direct and indirect effects and effects to the PCEs of critical habitat. The CTS-SC does not have a designated critical habitat.

5.2.10. California Clapper Rail

PCNB is expected to directly impact the CCR based on toxicity to avian species. Indirect effects from impacts on prey are also anticipated based on toxicity to birds, mammals, and possibly terrestrial invertebrates. Indirect effects from impacts on habitat are uncertain due to lack of toxicity data on aquatic and terrestrial plants.

Therefore, the Agency makes a **may affect, and likely to adversely affect** determination for the CCR. The CCR does not have a designated critical habitat.

5.2.11. California Freshwater Shrimp

PCNB is not expected to directly impact the CFWS based on toxicity to freshwater invertebrates based on daphnid toxicity data. However, there is significant uncertainty associated with this conclusion because mysid shrimp (estuarine/marine crustacean) are more sensitive to PCNB than daphnids (freshwater crustacean), and are more closely related to the CFWS. In addition, open-literature studies on freshwater benthic invertebrate species with PCB suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species.

Indirect effects from impacts on prey are not anticipated based on low toxicity to freshwater invertebrates. Indirect effects from impacts on habitat are uncertain due to lack of toxicity data for aquatic and terrestrial plants.

Based on the overall uncertainties for the CFWS, the Agency makes a **may affect, and likely to adversely affect** determination for this species. The CFWS does not have a designated critical habitat.

5.2.12. San Francisco Garter Snake

PCNB is expected to directly impact the SFGS based on toxicity to reptiles, using avian surrogate species data. Indirect effects from impacts on prey are also anticipated based on toxicity to terrestrial-phase amphibians, reptiles, mammals, and possibly terrestrial invertebrates. Indirect effects from impacts on habitat are anticipated due to effects on mammal burrow availability. SFGS rely on the burrows of small mammals for shelter and aestivation.

Therefore, the Agency makes a **may affect, and likely to adversely affect** determination for the SFGS. The SFGS does not have a designated critical habitat.

5.2.13. Delta Smelt and Tidewater Goby

PCNB is expected to directly impact the DS and TG based on toxicity to freshwater fish. Indirect effects from impacts on prey are also anticipated based on toxicity to estuarine/marine invertebrates. Indirect effects from impacts on habitat are uncertain due to lack of toxicity data for aquatic plants.

Therefore, the Agency makes a **may affect, and likely to adversely affect** determination for the DS and TG and a **habitat modification determination** for the designated critical habitat of the DS and TF based on the uncertainty of effects to the PCEs of critical habitat.

5.2.14. Addressing the Risk Hypotheses

In order to conclude this risk assessment, it is necessary to address the risk hypotheses defined in Section 2.9.1. Based on the conclusions of this assessment, two of the hypotheses can be rejected, meaning that three of the stated hypotheses represent concerns in terms of direct and indirect effects of PCNB on the BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and their designated critical habitat.

The labeled uses of PCNB may:

- directly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG by causing mortality or by adversely affecting growth or fecundity;
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of food supply;
- indirectly affect CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing aquatic habitat in their current range (via modification of water quality parameters, habitat morphology, and/or sedimentation);
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing terrestrial habitat in their current range (via reduction in small burrowing mammals leading to reduction in underground refugia/cover).

It is uncertain whether the labeled uses of PCNB are expected to:

- indirectly affect CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of the aquatic plant community in the species' current range, thus affecting primary productivity and/or cover;
- indirectly affect BCB, CTS-CC, CTS-SC, CTS-SB, DS, CCR, CFWS, SFGS, and TG and/or modify their designated critical habitat by reducing or changing the composition of the terrestrial plant community in the species' current range;

6. Uncertainties

General uncertainties that apply to most assessments completed for the San Francisco Bay Species Litigation are discussed in Attachment I. This section describes additional uncertainties specific to this assessment. Uncertainties listed below only apply to the currently marketed uses assessed in this document. However, as noted previously, there are other uses that are still technically registered as well as proposed uses that have not been assessed here. It is assumed that these additional uses will not alter the major risk conclusions of this assessment other than resulting in higher RQ values for different groups of organisms. The use of PCNB on sod farms would preclude the use of the Golf Course Adjustment Factor, thereby increasing the EECs for aquatic exposure.

6.1. Exposure Assessment Uncertainties

6.1.1. Terrestrial Exposure Assessment

6.1.1.a. T-REX

Organisms consume a variety of dietary items and may exist in a variety of sizes at different life stages. For foliar applications of liquid formulations, T-REX estimates exposure for the following dietary items: short grass, tall grass, broadleaf plants/small insects, fruits/pods/seeds/large insects, and seeds for granivores. Birds (used as a surrogate for amphibians and reptiles), including the CCR, and mammals, consume all of these items. The size classes of birds represented in T-REX are small (20 g), medium (100 g), and large (1000 g). The size classes for mammals are small (15 g), medium (35 g), and large (1000 g). EECs are calculated for the most sensitive dietary item and size class for birds (surrogate for amphibians and reptiles) and mammals. **Table 6-1** shows the percentages of the EECs and RQs of the various dietary classes for each size class as compared to the most sensitive dietary class (short grass) and size class (small mammal or bird). This information could be used to further characterize potential risk that is specific to the diets of birds and mammals. For example, if a mammal only consumes broadleaf plants and small insects and the RQ was 100 for small mammals consuming short grass, the RQ for small mammals that only consumed broadleaf plants and small insects would be 56 (100 x 0.56).

Table 6-1. Percentage of EEC or RQ for the Specified Dietary Items and Size Classes as Compared to the EEC or RQ for The Most Sensitive Dietary Items (Short Grass) and Size Class (Small Bird or Small Mammal)

Dietary Items	Percentage of EECs or RQs for the Specified Dietary Items and Size Class as compared to the EEC or RQ for Small Birds[1] or Small Mammals Consuming Short Grass					
Birds: Dose Based EECs and RQs						
Size Class	Small, 20 g		Mid, 100 g		Large, 1000 g	
	EEC	RQ	EEC	RQ	EEC	RQ
Short Grass	100%	100%	57%	45%	26%	14%
Tall Grass	46%	46%	26%	21%	12%	7%
Broadleaf plants/small Insects	56%	56%	32%	25%	14%	8%
Fruits/pods/seeds/large insects	6%	6%	4%	3%	2%	1%
Granivores	1%	1%	1%	1%	0.4%	0.2%
Mammals: Dose-Based EECs and RQs						
Size Class	Small, 15 g		Mid, 35 g		Large, 1000 g	
	EEC	RQ	EEC	RQ	EEC	RQ
Short Grass	100%	100%	69%	85%	16%	46%
Tall Grass	46%	46%	32%	39%	7%	21%
Broadleaf plants/small Insects	56%	56%	39%	48%	9%	26%
Fruits/pods/seeds/large insects	6%	6%	4%	5%	1%	3%
Granivores	1%	1%	1%	1%	0.2%	0.6%

Dietary Items	Percentage of EECs or RQs for the Specified Dietary Items and Size Class as compared to the EEC or RQ for Small Birds[1] or Small Mammals Consuming Short Grass
Mammals and Birds: Dietary-based EECs and RQs for all Size Classes[2]	
Short Grass	100%
Tall Grass	46%
Broadleaf plants/sm Insects	56%
Fruits/pods/seeds/lg insects	6%

[1] The percents of the maximum RQ shown here for birds are based on the Agency's default avian scaling factor of 1.15.
[2] Percentages for dose-based chronic EECs and RQs for mammals are equivalent to the acute dose-based EECs and RQs.

In the risk assessment, RQs were only calculated for the most sensitive dietary class relevant to the organisms assessed. For most organisms, not enough data is available to conclude that birds or mammals may not exclusively feed on a dietary class for at least some time period. However, most birds and mammals consume a variety of dietary items and thus the RQ will overestimate risk to those organisms. For example, the CCR is estimated to consume only 15% plant material (USFWS, 2003b). Additionally, some organisms will not feed on all of the dietary classes. For example, many amphibians would only consume insects and not any plant material.

6.1.1.b. T-HERPS

For foliar applications of liquid formulations, T-HERPS estimates exposure for the following dietary items: broadleaf plants/small insects, fruits/pods/seeds/large insects, small herbivore mammals, small insectivore mammals, and small amphibians. Snakes and amphibians may consume all of these items. The default size classes of amphibians represented in T-HERPS are small (2 g), medium (20 g), and large (200 g). The default vertebrate prey size that the medium and large amphibians can consume is 13 g and 133 g, respectively (small amphibians are not expected to eat vertebrate prey). The default size classes for snakes are small (2 g), medium (20 g), and large (800 g). The default vertebrate prey size that medium and large snakes can consume is 25 g and 1,286 g, respectively (small snakes are not expected to eat vertebrate prey). EECs are calculated for the most sensitive dietary item and size class for amphibians and snakes. **Table 6-2** shows the percentages of the EECs and RQs of the various dietary classes for each size class as compared to the most sensitive dietary class (herbivorous mammal) and size class [medium (20 g) amphibian or snake]. This information could be used to further characterize potential risk that is specific to the diet of amphibians and snakes.

Table 6-2. Percentage of EEC or RQ for the Specified Dietary Class as Compared to the EEC or RQ for The Most Sensitive Dietary Class (Small Herbivore Mammals) and Size Class (Medium Amphibian or Snake)

Dietary Items	Percentage of EECs or RQs for the Specified Dietary Items and Size Class as compared to the EEC or RQ for Medium Amphibians or Snakes Consuming Small Herbivore Mammals		
Amphibians: Acute Dose Based EECs and RQs			
Size Class	Small, 2 g	Mid, 20 g	Large, 200 g
Broadleaf plants/sm Insects	5%	3%	2%
Fruits/pods/seeds/lg insects	0.5%	0.3%	0.2%

Small herbivore mammals	N/A	100%	37%	
Small insectivore mammals	N/A	6%	2%	
Small amphibians	N/A	2%	1%	
Snakes: Acute Dose-Based EECs and RQs				
Size Class	Small, 2 g	Mid, 20 g	Mid, 200 g[1]	Large, 800 g
Broadleaf plants/sm Insects	3%	2%	1%	1%
Fruits/pods/seeds/lg insects	0.4%	0.2%	0.1%	0.1%
Small herbivore mammals	N/A	100%	40%	23%
Small insectivore mammals	N/A	6%	3%	1%
Small amphibians	N/A	2%	2%	1%
Amphibians and Snakes: Acute and Chronic Dietary-based EECs and RQs for all Size Classes				
	Amphibians		Snakes	
Broadleaf plants/sm Insects	56%		73%	
Fruits/pods/seeds/lg insects	6%		8%	
Small herbivore mammals	100%		100%	
Small insectivore mammals	6%		6%	
Small amphibians	2%		2%	

[1] To provide more information, a 200 g snake (eating a 291 g prey item) was also modeled (in addition to the default body sizes).

In the risk assessment, RQs were only calculated for the most sensitive dietary class relevant to the organisms assessed. For most organisms, not enough data are available to conclude that amphibians or snakes may not exclusively feed on a dietary class for at least some time period. However, most amphibians and snakes consume a variety of dietary items and thus the RQ will overestimate risk to those organisms. Additionally, some organisms will not feed on all of the dietary classes. For example, many amphibians would only consume insects and not any plant material.

6.1.2. Exposure in Estuarine/Marine Environments

PRZM-EXAMS modeled EECs are intended to represent exposure of aquatic organisms in relatively small ponds and low-order streams. Therefore it is likely that EECs generated from the PRZM-EXAMS model will over-estimate potential concentrations in larger receiving water bodies such as estuaries, embayments, and coastal marine areas because chemicals in runoff water (or spray drift, etc.) should be diluted by a much larger volume of water than would be found in the ‘typical’ EXAMS pond. However, as chemical constituents in water draining from freshwater streams encounter brackish or other near-marine-associated conditions, there is potential for important chemical transformations to occur. Many chemical compounds can undergo changes in mobility, toxicity, or persistence when changes in pH, Eh (redox potential), salinity, dissolved oxygen (DO) content, or temperature are encountered. For example, desorption and re-mobilization of some chemicals from sediments can occur with changes in salinity (Jordan *et al.,* 2008; Means, 1995; Swarzenski *et al.,* 2003), changes in pH (*e.g*., Wood and Baptista 1993), Eh changes (Velde and Church, 1999; Wood and Baptista, 1993), and other factors. Thus, although chemicals in discharging rivers may be diluted by large volumes of water within receiving estuaries and embayments, the hydrochemistry of the marine-influenced water may negate some of the attenuating impact of the greater water volume; for example, the effect of dilution may be confounded by changes in chemical mobility (and/or bioavailability) in brackish water. In addition, freshwater contributions from discharging streams and rivers do not instantaneously mix with more saline water bodies. In these settings, water will commonly remain highly stratified, with fresh water lying atop denser, heavier saline water – meaning that exposure to concentrations found in discharging stream water may propagate some distance beyond the outflow point of the stream (especially near the water surface). Therefore, it is not assumed that discharging water will be rapidly diluted by the entire water volume within an estuary, embayment, or other coastal aquatic environment. PRZM-EXAMS model results should be considered consistent with concentrations that might be found near the head of an estuary unless there is specific information – such as monitoring data – to indicate otherwise. Conditions nearer to the mouth of a bay or estuary, however, may be closer to a marine-type system, and thus more subject to the notable buffering, mixing, and diluting capacities of an open marine environment. Conversely, tidal effects (pressure waves) can propagate much further upstream than the actual estuarine water, so discharging river water may become temporarily partially impounded near the mouth (discharge point) of a channel, and resistant to mixing until tidal forces are reversed.

The Agency does not currently have sufficient information regarding the hydrology and hydrochemistry of estuarine aquatic habitats to develop alternate scenarios for assessed listed species that inhabit these types of ecosystems. The Agency acknowledges that there are unique brackish and estuarine habitats that may not be accurately captured by PRZM-EXAMS modeling results, and may, therefore, under- or over-estimate exposure, depending on the aforementioned variables.

For PCNB, there is uncertainty regarding the extent to which PCNB is subject to biotic degradation in both the terrestrial and aquatic environments. Aerobic soil metabolism data are limited, and the anaerobic soil metabolism data are of questionable value due to limited data points and limitations of the experimental method used. There is additional uncertainty due to the lack of aerobic and anaerobic aquatic metabolism data. This uncertainty has been accounted for by using twice the respective soil metabolism half-lives as a surrogate for the aquatic metabolism half-life modeling input values. The RED characterized the parent as likely to biodegrade slowly under aerobic conditions, but noted that PCNB was metabolized more rapidly under anaerobic soil conditions.

The lack of data for individual degradates also adds to the uncertainty, as a total residues approach was used for determining half-lives, yet half-lives for photodegradation and hydrolysis were for parent PCNB only. Data on the photodegradation of the degradate PCA in water are not available. In the absence of these data, EFED has assumed that, in general, PCNB would degrade to PCA in soil and that the PCA would then remain stable to further degradation in both the aquatic and terrestrial environments. Data on the photolysis in air of both PCNB and PCA are not available. Given that PCA is the major degradate remaining after microbial degradation, and is a volatile compound, photodegradation in air data are important for both PCNB and PCA to more accurately characterize the extent to which either PCNB and/or PCA will persist in the atmospheric compartment and be available for long-range transport. In the absence of these data, EFED has presumed that PCNB is likely to volatilize from applications where it is not incorporated into soil, that it will be stable to degradation in the atmosphere, and that it will be subject to long-range atmospheric transport. Similar assumptions are made for PCA. These assumptions could result in overstating the potential for long-range transport of PCNB and PCA.

6.2. Effects Assessment Uncertainties

6.2.1. PCNB and Degradate Data Gaps and Uncertainties

No data were available on the effects of PCNB on either aquatic or terrestrial plants; therefore, it is not possible to assess the risks to plants associated with the current uses of PCNB. However, the only two incidents reported for PCNB affected terrestrial plants. Open literature suggests that, like aquatic animals, aquatic plants are able to accumulate PCNB residues and serve as a route of entry for PCNB into the aquatic food chain, although compounds such as PCNB and PCA are associated with relatively low dietary absorption efficiency and short metabolic half-lives. However, the likely persistence and toxicity of PCNB and its degradates coupled with the chemicals' propensities to bioconcentrate, are concerns that underscore the potential vulnerability of aquatic communities to PCNB exposure resulting from labeled uses of the fungicide.

Data on the foliar dissipation half-life of PCNB are not available and therefore the terrestrial exposure portion of this assessment used the default value of 35 days. The only use where a dissipation rate was used to calculate EECs was for turf foliar applications. In this case, even lowering the foliar dissipation half life from 35 days to 1 day would not change whether or not LOCs were exceeded for birds (and reptiles and amphibians) or mammals. Therefore, the lack of foliar dissipation data is considered a minor uncertainty.

Data on the toxicity of PCNB to sediment-dwelling invertebrates are not available. The likely presence of PCNB in benthic sediments could serve as a route of entry into aquatic food chains. In addition, since benthic invertebrates can serve as an important food source for some of the species in this assessment (CCR and TG), the lack of understanding as to the impact of PCNB on benthic organisms is considered an uncertainty.

6.2.2. Uncertainties from Potential Dioxin Exposure and Toxicity

Exposure and toxicity to PCDDs and PCDFs released as a result of application of PCNB were not evaluated in the risk estimation portion of this assessment. However, based on their environmental fate and effects properties, there may be the potential for dioxins to directly or indirectly impair the listed species in this assessment; therefore, discussion of the characteristics of dioxins and their possible impact on this assessment is warranted.

Dioxins are listed under the Stockholm Convention as Persistent Organic Pollutants (POPs). A great deal of research has been conducted to characterize the toxicities of PCDDs and PCDFs. In addition, on October 15, 2004, the EPA transmitted to the National Academy of Sciences (NAS) the NAS Review Draft of EPA's Exposure and Human Health Reassessment of 2,3,7,8-

Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds (USEPA, 2003a)[12]. General fate and effects information from this and other reports/studies is discussed here.

In terms of fate properties, dioxins are highly lipophilic and have low water solubility, so they are highly immobile in soil. When released to land, such as through application of pesticides contaminated with dioxins, dioxins will sorb to soil particles and soil organic matter. If they reach surface water through runoff and/or erosion of soil, dioxins will remain sorbed to particulate matter or will partition to suspended solids or dissolved organic matter in the water column. Dioxins may be taken up by biota from the water column, but once sorbed to organic matter or suspended particles will tend to remain as such, eventually undergoing sedimentation and burial. Dioxins have been detected throughout the environment, with concentrations that vary with depth in soil and sediment, demonstrating their presence and persistence over time. Their extremely high tendency for sorption to soil/sediment increases their persistence by making them less available for photodegradation or biotic degradation. Once in the environment, dioxins tend to end up in soil or sediment, or taken up by biota. It should be noted that all of listed species in this assessment, except for the BCB, either spend a portion of their lives in the benthic environment or feed on organisms that may live or feed near sediment.

Aquatic organisms may bioaccumulate dioxins from aquatic sediments and may bioconcentrate dioxins present in the water column. The majority of uptake by biota is likely "*primarily food chain-based starting with uptake by benthic* [sediment dwelling] *organisms (e.g., mussels, chironomids) directly from sediment pore waters and/or by ingestion or filtering of contaminated particles* (USEPA, 2003a)." Terrestrial organisms may also bioaccumulate dioxins, particularly through the consumption of contaminated food. The primary way dioxins enter the terrestrial food chain is by ingestion of plants onto which dioxins have been deposited through wet or vapor phase deposition (or less likely through dry or particulate phase deposition). For dioxins introduced into the environment via applications of PCNB, a more likely means of entering the terrestrial food chain would be through direct ingestion of PCNB residues through *"soil ingestion by earthworms, fur preening by burrowing animals, incidental ingestion by grazing animals, etc* (USEPA, 2003a)." Additionally, dioxins may enter the terrestrial food chain by ingestion of plants contaminated during foliar application of PCNB. In general, considering the Log K_{ow} values of PCDDs and PCDFs (Log K_{ow} range: 7.9-9.5) relative to those of PCNB and its degradates (5.0; see USEPA, 2010 for further information on how this value was derived), these contaminants would be expected to bioaccumulate to a greater extent than PCNB and its degradates.

In terms of effects, demonstrated toxicity of 2,3,7,8-TCDD and other similar-acting PCDD, PCDF and PCB congeners in fish, birds, and mammals include adverse effects on reproduction, development, and endocrine functions, wasting syndrome, immunotoxicity, and mortality in both laboratory and field settings (USEPA, 2008). Specific effects recorded in fish larvae exposed to 2,3,7,8-TCDD include pericardial, yolk sac, and meningeal edema, impaired jaw development, impaired heart development and function, reduced trunk blood flow, anemia, hemorrhage, growth retardation, and mortality (USEPA, 2008). Exposure of birds to dioxins leads to mortality, deformity and inhibited development in offspring (USEPA, 2003b). Dioxins are not metabolized by mammals (Van Den Berg *et al.,* 2006)

[12] Available at http://www.epa.gov/ncea/pdfs/dioxin/nas-review

Based solely on a qualitative examination of the environmental fate and effects properties of dioxins, bioaccumulation of dioxins in aquatic organisms could adversely impact CCR, SFGS, CTS, TG, or DS through either direct consumption of contaminated prey or through indirect impacts to birds, mammals, or fish species that serve as prey or influence habitat of these listed species. Bioaccumulation, in combination with known toxicity to birds, mammals, and fish could lead to risks of concern with sufficient levels of exposure; however, without a quantitative analysis linking exposure and effects, potential affects to the listed species in this assessment is uncertain.

6.2.3. Use of Surrogate Species Effects Data

Guideline toxicity tests and open literature data on PCNB are not available for reptiles or terrestrial-phase amphibians; therefore, birds are used as surrogate species for reptiles, terrestrial-phase amphibians, the SFGS, and the CTS. Reptiles and amphibians are poikilotherms (body temperature varies with environmental temperature) while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). Therefore, reptiles and amphibians tend to have much lower metabolic rates and lower caloric intake requirements than birds or mammals. As a consequence, birds are likely to consume more food than amphibians or reptiles. Consequently, use of avian food intake allometric equation as a surrogate for reptile and terrestrial-amphibians is likely to result in an over-estimation of exposure. Therefore, endpoints based on bird ecotoxicity data are assumed to be protective of potential direct effects to reptiles and terrestrial-phase amphibians including the SFGS and CTS, and extrapolation of the risk conclusions from the most sensitive tested species to the SFGS and CTS are likely to overestimate the potential risks to those species. The T-HERPS model attempts to account for this difference and refines the risk estimation for reptiles and terrestrial-phase amphibians.

No data are available in the open literature on the effects of PCNB or its degradates to lepidopterans, and EPA does not require registrants to submit data on this taxon. In this assessment, bees are used as surrogates for lepidopterans.

Efforts are made to select the organisms most likely to be affected by the type of compound and usage pattern; however, there is an inherent uncertainty in extrapolating across phyla. In addition, the Agency's LOCs are intentionally set very low, and conservative estimates are made in the screening level risk assessment to account for these uncertainties.

6.2.4. Sublethal Effects

When assessing acute risk, the screening risk assessment relies on the acute mortality endpoint as well as a suite of sublethal responses to the pesticide, as determined by the testing of species response to chronic exposure conditions and subsequent chronic risk assessment. Consideration of additional sublethal data in the effects determination is exercised on a case-by-case basis and only after careful consideration of the nature of the sublethal effect measured and the extent and quality of available data to support establishing a plausible relationship between the measure of effect (sublethal endpoint) and the assessment endpoints. However, the full suite of sublethal effects from valid open literature studies is considered for the characterization purposes.

To the extent to which sublethal effects are not considered in this assessment, the potential direct and indirect effects of PCNB on listed species may be underestimated.

6.2.5. Acute LOC Assumptions

The risk characterization section of this assessment includes an evaluation of the potential for individual effects. The individual effects probability associated with the acute RQ is based on the assumption that the dose-response curve fits a probit model. It uses the mean estimate of the slope and the LC_{50} to estimate the probability of individual effects. When raw data associated with the LC_{50} is not available, a default value of 4.5 is used for the probit slope.

7. Risk Conclusions

In fulfilling its obligations under Section 7(a)(2) of the Endangered Species Act, the information presented in this endangered species risk assessment represents the best data currently available to assess the potential risks of PCNB to BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS and their designated critical habitat.

Based on the best available information, the Agency makes a Likely to Adversely Affect determination for the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS. Additionally, the Agency has determined that there is the potential for modification of the designated critical habitat for the BCB, DS, TG, CTS-CC, and CTS-SB from the use of the chemical. Given the LAA and potential modification of designated critical habitat determinations, a description of the baseline status and cumulative effects is provided in Attachment III.

A summary of the risk conclusions and effects determinations for the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS and their critical habitat, given the uncertainties discussed in Section 6 and Attachment I, is presented in **Table 7-1**. Use specific effects determinations are provided in **Table 7-2** and **Table 7-3**.

In this document, only currently marketed uses of PCNB on cole crops, cotton, potatoes, and turf were assessed. Since LAA determinations were made for all seven listed species evaluated, the inclusion of additional uses of PCNB are not expected to affect the overall outcome of this assessment. However, due to differences in exposure modeling and other analysis factors associated with individual agricultural and non-agricultural uses, it is not possible to predict which additional uses of PCNB, if marketed, would result in exceedances of Agency LOCs for direct and indirect effects to taxa linked to the species in this assessment.

Table 7-1. Effects Determination Summary for Effects of PCNB on the BCB, CTS-CC, CTS-SC, CTS-SB, CCR, CFWS, SFGS, TG, and DS

Species	Effects Determination	Basis for Determination
Bay Checkerspot		**Potential for Direct Effects**

Species	Effects Determination	Basis for Determination
Butterfly (*Euphydryas editha bayensis*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	***Terrestrial*** Acute and chronic RQ values could not be calculated for direct effects to the BCB because there is no definitive acute toxicity endpoint for the honeybee (surrogate for BCB in this assessment) and no relevant chronic terrestrial insect data from the open literature. If EECs for arthropods were compared to the highest dose tested in the acute contact honeybee toxicity study (100 µg/bee converted to 781.25 µg/g based on assumed weight of 0.128 g/bee), the ratio would range from 0.24 to 6.3 for the various PCNB uses, indicating that there is some uncertainty surrounding risk to the BCB based on lack of definitive surrogate toxicity data. In order for there not to be risks of concern to the BCB, the definitive honeybee LD_{50} would have to be ≥12,503 µg/bee to not exceed the terrestrial invertebrate listed species LOC of 0.05; this is approximately one-tenth the weight of an adult honeybee. Therefore, due to the relatively high application rates of PCNB, there is significant uncertainty in the potential for effects to the BCB based on available toxicity information. Given the uncertainty in the risk to the BCB based on available honeybee data, combined with the lack of toxicity data on the effects of PCNB on lepidopterans, there is insufficient effects information to determine potential direct effects to the BCB for all PCNB uses evaluated in this assessment.
		Potential for Indirect Effects
		Terrestrial food items, habitat The BCB relies on terrestrial plants exclusively for both food and habitat and has an obligate relationship with dicots. However, no relevant terrestrial plant toxicity data have been identified for PCNB or its degradates. The only two incidents reported for PCNB affected terrestrial plants. Due to the lack of effects data on terrestrial plants, there is insufficient information to determine potential indirect effects to the BCB for all PCNB uses evaluated in this assessment.
California Tiger Salamander (All 3 DPS) (*Ambystoma californiense)*	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Aquatic-phase (Eggs, Larvae, and Adults) Acute RQ values based on freshwater fish (surrogate for aquatic-phase amphibians) toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the CTS for those uses. In addition, individual effect probabilities for CTS based on freshwater fish data range from 1-in-9.1 to 1-in-2.8×10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the aquatic-phase CTS as a result of cole crop and turf uses of PCNB.
		Terrestrial-phase (Juveniles and Adults) The chronic risk LOC for birds (surrogate for terrestrial-phase CTS) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, when worms are considered as a dietary item for the CTS, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CTS, indicating the potential for chronic risk to the CTS for these crop uses of PCNB based on worm dietary items. Acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (surrogate for terrestrial-phase amphibians). Therefore, there is a potential for direct effects to the

Species	Effects Determination	Basis for Determination
		terrestrial-phase CTS for all PCNB uses evaluated in this assessment.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover, and primary productivity RQs for freshwater fish (surrogate for aquatic-phase amphibian dietary items) only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the CTS based on this prey component (*i.e.*, non-listed aquatic-phase amphibians) is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are $\leq$1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the CTS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. RQs cannot be calculated for aquatic plants due to a lack of toxicity data, representing a source of uncertainty. These results suggest that there is little potential for any of the uses of PCNB to affect the prey-base of the aquatic-phase CTS, but effects on its habitat are uncertain due to lack of toxicity data on aquatic plants.
		Terrestrial prey items, habitat As described for the BCB above, there is uncertainty in potential effects to terrestrial invertebrates (prey item) for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB. Based on submitted rat two-generation reproductive toxicity data, the chronic risk LOC for mammals is exceeded for all PCNB uses, indicating that small mammal prey may be adversely affected by PCNB at current use rates. EECs for mammals feeding on short grass would have to be up to 520 times lower to alleviate concerns of chronic effects to small mammals. Therefore, there is the potential for indirect effects to CTS habitat based on chronic risk to small mammals. Based on these results, there is the potential for all uses of PCNB evaluated in this assessment to indirectly affect the terrestrial-phase CTS via terrestrial invertebrate prey as well as habitat in small mammal burrows.
California Clapper Rail (*Rallus longirostris obsoletus*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Terrestrial Based on avian reproductive toxicity data, the chronic risk LOC is exceeded for all PCNB uses except for cotton. EECs for small birds feeding on short grass would have to be up to 21 times lower to alleviate concerns of direct chronic effects to the CCR. In addition, when worms are considered as a dietary item for the CCR, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the CCR, indicating the potential for chronic risk to the CCR for these crop uses of PCNB based on worm dietary items. Acute effects cannot be precluded for any PCNB uses because there is a

Species	Effects Determination	Basis for Determination
		non-definitive endpoint for acute toxicity to birds. Therefore, there is the potential for direct effects to the CCR for all PCNB uses evaluated in this assessment.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover, and primary productivity Freshwater fish RQs only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the CCR for all PCNB uses based on this prey component is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the CCR based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. No acute or chronic RQ values representing any uses of PCNB exceed LOCs for estuarine/marine fish. The probability of an individual effect for estuarine/marine fish ranges from 1-in-$9.3x10^{4}$ to 1-in-$3.5x10^{6}$ across PCNB uses. Therefore, there is low potential for indirect effects to the CCR based on this prey component. Estuarine/marine invertebrate acute RQ values exceed the non-listed species LOC (0.5) for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, indirect effects to the CCR are anticipated based on this prey component. These results show that PCNB is likely to indirectly affect the CCR via freshwater fish and estuarine/marine invertebrate prey under exposure scenarios evaluated in this assessment.
		Terrestrial prey items, riparian habitat As describe for direct effects to the CCR above, there is the potential for chronic effect to birds (as prey for CCR) for all PCNB uses except for cotton. In addition, the possibility of acute affects to birds cannot be precluded for any of the PCNB uses based on available data. As described for the indirect effects to the CTS above, the mammalian chronic risk LOC is exceeded for all PCNB uses, indicating that there is potential for indirect effects to the CCR based on potential effects to small mammal prey items. As described for the BCB above, there is uncertainty in potential effects to terrestrial invertebrates for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB. No relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is not possible to determine indirect effects to the CCR based on this prey item. Based on these results, there is the potential for all uses of PCNB to

Species	Effects Determination	Basis for Determination
		indirectly affect the CCR via effects on bird, mammal, and possibly invertebrate prey.
California Freshwater Shrimp (*Syncaris pacifica*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects** ***Aquatic*** Acute RQs do not exceed the acute risk to non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates range from 1-in-1x10^{11} to 1-in-1x10^{16} across PCNB uses. Therefore, direct effects are not expected to the CFWS based on risk to freshwater invertebrate prey under the exposure scenarios evaluated in this assessment and based on available toxicity data. It should be noted that there is significant uncertainty associated with this conclusion because of the higher toxicity of PCNB to mysid shrimp (estuarine/marine crustacean) compared to *Daphnia magna* (freshwater crustacean). Mysid shrimp are more closely related to the CFWS, and may indicate that effects to the CFWS may be underestimated based on the toxicity endpoint from *D. magna*. In addition, open-literature studies on freshwater benthic invertebrate species with pentachlorobenzene (a degradate of PCNB) reported acute LC_{50} values ranging from 51 to 230 µg/L, which are at least three times lower than the daphnid acute endpoint (770 µg/L) used to calculate RQs for freshwater invertebrates in this assessment. Although studies for all three species were only deemed useful for qualitative purposes, it does suggest that *D. magna* may not be as sensitive to PCNB or its degradates as other species. Moreover, if any of the other freshwater invertebrate species were used to calculate RQs for the CFWS, it would result in exceedance of the acute risk to listed species LOC. Based on toxicity data from freshwater invertebrates, there is low potential for direct effects to the CFWS as a result of PCNB uses. However, there is uncertainty associated with this conclusion based on higher toxicity of PCNB to mysid shrimp and several freshwater invertebrates and the potential for bioaccumulation of PCNB in CFWS aquatic dietary items.
		Potential for Indirect Effects ***Aquatic prey items, aquatic habitat, cover, and primary productivity*** As described for the direct effects to the CFWS above, there is low potential for direct effects to freshwater inverterbrates as a result of PCNB uses. However, there is uncertainty associated with this conclusion based on higher toxicity of PCNB to mysid shrimp and several freshwater invertebrates and the potential for bioaccumulation of PCNB in CFWS aquatic dietary items. RQs cannot be calculated for aquatic plants due to a lack of toxicity data, representing a source of uncertainty concerning the effects of PCNB on plant food sources and habitat. These results suggest there is low potential for any uses of PCNB to indirectly impact the freshwater invertebrate prey-base of the CFWS. However, there is significant uncertainty associated with this finding due to lack of toxicity data on aquatic plants and the potential underestimation of toxicity for aquatic invertebrates.

Species	Effects Determination	Basis for Determination
		Terrestrial prey items, riparian habitat No relevant data on terrestrial plants have been identified for PCNB or its degradates; therefore, it is uncertain whether PCNB is likely to indirectly affect the CFWS via terrestrial plant food and habitat.
San Francisco Garter Snake (*Thamnophis sirtalis tetrataenia*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Terrestrial The chronic risk LOC for birds (as a surrogate for SFGS) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, when worms are considered as a dietary item for the SFGS, estimated worm residues from potato and cole crops uses are greater than the avian chronic dietary endpoint value of 600 mg/kg-diet used to evaluate chronic risk to the SFGS, indicating the potential for chronic risk to the SFGS for these crop uses of PCNB based on worm dietary items. Acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (surrogate for SFS). Therefore, there is a potential for direct effects to the SFGS for all PCNB uses evaluated in this assessment.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover, and primary productivity Freshwater fish (and aquatic-phase amphibian) RQs only exceed the acute risk to listed species LOC for cole crop and turf uses of PCNB, indicating that the potential for indirect effects to the SFGS for all PCNB uses based on this prey component is low. For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the SFGS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. These results suggest that there is low potential for any PCNB use to indirectly affect the SFGS via freshwater fish and amphibian prey under exposure scenarios evaluated in this assessment.
		Terrestrial prey items, riparian habitat The chronic risk LOC for birds (as a surrogate for terrestrial-phase amphibians and reptiles) is exceeded for all PCNB uses except for cotton, and remains above the LOC after refinement using the T-HERPS model. In addition, acute effects cannot be precluded for any PCNB use because there is a non-definitive endpoint for acute toxicity to birds (as a surrogate for terrestrial-phase amphibians and reptiles). Therefore, there is a potential for indirect effects to the SFGS for all PCNB uses evaluated in this assessment. As described for the indirect effects to the CTS above, the mammalian chronic risk LOC is exceeded for all PCNB uses, indicating that there is potential for indirect effects to the SFGS based on potential effects to small mammal prey items. As described for the BCB above, there is uncertainty in potential effects to

Species	Effects Determination	Basis for Determination
		terrestrial invertebrates (prey item) for all PCNB uses due to the non-definitive endpoint for honeybees and the relatively high application rates of PCNB. Based on these results, there is the potential for PCNB to indirectly affect the SFGS via reptile, amphibian, mammal, and possibly invertebrate prey as well as the mammal burrow component of habitat.
Delta Smelt (*Hypomesus transpacificus*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Aquatic Acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the DS for those uses. In addition, individual effect probabilities for DS based on freshwater fish data range from 1-in-9.1 to 1-in-2.8x10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the DS as a result of cole crop and turf uses of PCNB.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-1x10^{11} across PCNB uses. Therefore, there is low potential for indirect effects to the DS based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. Conversely, for estuarine/marine invertebrates, acute RQ values exceed the non-listed species LOC (0.5) for estuarine/marine invertebrates for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, there is the potential for indirect effects to the DS based on this prey component. RQs cannot be calculated for terrestrial or aquatic plants due to a lack of toxicity data, representing a source of uncertainty. Based on these results, PCNB is likely to indirectly affect the DS via estuarine/marine invertebrate prey, and possibly terrestrial and aquatic plant habitat, under exposure scenarios evaluated in this assessment.
Tidewater Goby (*Eucyclogobius newberryi*)	May Affect, Likely to Adversely Affect (LAA) [Applies to cole crop, cotton, potato, and turf uses of PCNB]	**Potential for Direct Effects**
		Aquatic Acute RQ values based on freshwater fish toxicity data exceed the acute risk to listed species LOC for for cole crop and turf uses of PCNB, indicating the potential for direct acute risks to the TG for those uses. In addition, individual effect probabilities for TG based on freshwater fish data range from 1-in-9.1 to 1-in-2.8x10^{11} across PCNB uses. Therefore, there is a potential for direct effects to the DS as a result of cole crop and turf uses of PCNB.
		Potential for Indirect Effects
		aquatic prey items, terrestrial/aquatic habitat

Species	Effects Determination	Basis for Determination
		For freshwater invertebrates, RQs do not exceed the acute risk non-listed species LOC (0.5) or chronic LOC (1) for direct effects to freshwater invertebrates for any PCNB uses. Individual effect probabilities for freshwater invertebrates are ≤1-in-$1x10^{11}$ across PCNB uses. Therefore, there is low potential for indirect effects to the TG based on risk to freshwater invertebrate prey for all PCNB uses evaluated in this assessment. Conversely, for estuarine/marine invertebrates, acute RQ values exceed the non-listed species LOC (0.5) for estuarine/marine invertebrates for cole crop and turf uses of PCNB, while chronic RQ values exceed the chronic LOC (1) for all current uses of PCNB. Peak and 21-day EECs would have to be approximately five and 43 times lower to alleviate risks of concern to estuarine/marine prey organisms based on acute and chronic toxicity data, respectively. Therefore, there is the potential for indirect effects to the TG based on this prey component. RQs cannot be calculated for terrestrial or aquatic plants due to a lack of toxicity data, representing a source of uncertainty. Based on these results, PCNB is likely to indirectly affect the TG via estuarine/marine invertebrate prey, and possibly terrestrial and aquatic plant habitat, under exposure scenarios evaluated in this assessment.

Table 7-2. Use Specific Summary of the Potential for Adverse Effects to Aquatic Taxa

Uses	Potential for Effects to Identified Taxa Found in the Aquatic Environment:									
	Estuarine/Marine Vertebrates[1]		Tidewater Goby, Delta Smelt, California Tiger Salamander (all DPS) and Freshwater Vertebrates[2]		California Freshwater Shrimp and Freshwater Invertebrates[3]		Estuarine/Marine Invertebrates[4]		Vascular Plants[5]	Non-vascular Plants[5]
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic		
Cole crops	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC
Cotton	No	No	No	No	No	No	No	Yes	UNC	UNC
Potatoes	No	No	No	No	No	No	No	Yes	UNC	UNC
Turf (foliar)	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC
Turf (granular)	No	No	Yes*	No	No	No	Yes	Yes	UNC	UNC

UNC = uncertain due to lack of effects data or non-definitive toxicity data where risks of concern cannot be precluded
DPS = distinct population segments
[1] A yes in this column indicates a potential for indirect effects to CCR.
[2] A yes in this column indicates a potential for direct effects to the TG, DS, CTS-CC, CTS-SC, and CTS-SB, and potential for indirect effects to SFGS, CCR, CTS-CC, CTS-SC, and CTS-SB.
[3] A yes in this column indicates a potential for direct effects to the CFWS and indirect effects to the CFWS, TG, DS, SFGS, CCR, CTS-CC, CTS-SB, CTS-SC.
[4] A yes in this column indicates a potential for indirect effects to TG, DS, CCR.
[5] A yes in this column indicates a potential for indirect effects to SFGS, CCR, TG, DS, CTS-CC, CTS-SC, CTS-SB, and CFWS.
* RQ exceeds the LOC for listed species (potential for direct effects) but not for non-listed species (no potential for indirect effects).

Table 7-3. Use Specific Summary of the Potential for Adverse Effects to Terrestrial Taxa

Uses	Potential for Effects to Identified Taxa Found in the Terrestrial Environment:										
	Small Mammals[1]		California Clapper Rail and Small Birds[2]		California Tiger Salamander (all DPS) and Amphibians[3]		San Francisco Garter Snake and Reptiles[4]		Bay Checkerspot Butterfly and Terrestrial Invertebrates (Acute)[5]	Dicots[6]	Monocots[6]
	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic			
Cole crops	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Cotton	No	Yes	UNC	No	UNC	No	UNC	No	UNC*	UNC	UNC
Potatoes	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Turf (foliar)	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC
Turf (granular)	No	Yes	UNC	Yes	UNC	Yes	UNC	Yes	UNC*	UNC	UNC

UNC = uncertain due to lack of effects data or non-definitive toxicity data where risks of concern cannot be precluded
DPS = distinct population segments
[1] A yes in this column indicates a potential for indirect effects to SFGS, CCR, CTS-CC, CTS-SC, CTS, and CTS-SB.
[2] A yes in this column indicates a potential for direct effects to CCR and indirect effects to the CCR, SFGS, CTS-CC, CTS-SC, and CTS-SB.

[3] A yes in this column indicates a potential for direct CTS-CC, CTS-SC, CTS-SB, and indirect effects to CTS-CC, CTS-SC, CTS-SB, SFGS, and CCR.
[4] A yes in this column indicates the potential for direct and indirect effects to SFGS and other reptiles.
[5] A yes in this column indicates a potential for direct effect to BCB and indirect effects to SFGS, CCR, CTS-CC, CTS-SC, and CTS-SB.
[6] A yes in this column indicates a potential for indirect effects to BCB, SFGS, CCR, TG, DS, CTS-CC, CTS-SC, CTS-SB, and CFWS.
* There is some indication that PCNB use on cole crops could adversely impact earthworms resulting in indirect effects to the CTS, CCR, and SFGS, which could prey on worms (see Section 5.2.2.b for details).

Based on the conclusions of this assessment, a formal consultation with the U. S. Fish and Wildlife Service under Section 7 of the Endangered Species Act should be initiated.

When evaluating the significance of this risk assessment's direct/indirect and adverse habitat modification effects determinations, it is important to note that pesticide exposures and predicted risks to the listed species and its resources (*i.e.*, food and habitat) are not expected to be uniform across the action area. In fact, given the assumptions of drift and downstream transport (*i.e.*, attenuation with distance), pesticide exposure and associated risks to the species and its resources are expected to decrease with increasing distance away from the treated field or site of application. Evaluation of the implication of this non-uniform distribution of risk to the species would require information and assessment techniques that are not currently available. Examples of such information and methodology required for this type of analysis would include the following:

- Enhanced information on the density and distribution of BCB, CTS (all DPS), CCR, CFWS, TG, DS, and SFGS life stages within the action area and/or applicable designated critical habitat. This information would allow for quantitative extrapolation of the present risk assessment's predictions of individual effects to the proportion of the population extant within geographical areas where those effects are predicted. Furthermore, such population information would allow for a more comprehensive evaluation of the significance of potential resource impairment to individuals of the assessed species.
- Quantitative information on prey base requirements for the assessed species. While existing information provides a preliminary picture of the types of food sources utilized by the assessed species, it does not establish minimal requirements to sustain healthy individuals at varying life stages. Such information could be used to establish biologically relevant thresholds of effects on the prey base, and ultimately establish geographical limits to those effects. This information could be used together with the density data discussed above to characterize the likelihood of adverse effects to individuals.
- Information on population responses of prey base organisms to the pesticide. Currently, methodologies are limited to predicting exposures and likely levels of direct mortality, growth or reproductive impairment immediately following exposure to the pesticide. The degree to which repeated exposure events and the inherent demographic characteristics of the prey population play into the extent to which prey resources may recover is not predictable. An enhanced understanding of long-term prey responses to pesticide exposure would allow for a more refined determination of the magnitude and duration of resource impairment, and together with the information described above, a more complete prediction of effects to individual species and potential modification to critical habitat.

8. References

A complete list of submitted studies with MRIDs for PCNB are provided in Appendix J.

A bibliography of ECOTOX references, identified by the letter E followed by a number, is located in Appendix G.

Arnot, J. A., & Gobas, F. A. P. C. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environmental Toxicology and Chemistry, 23*(10), 2343-2355.

Beck, J. and K.E. Hansen. 1974. The degradation of quintozene, pentachlorobenzene, hexachlorobenzene and pentachloroaniline in soil. Pestic. Sci. 5:41-48.

Brooke L. 1987. Report of the Flow-Through and Static Acute Test Comparisons with Fathead Minnows and Acute Tests with an Amphipod and a Cladoceran. "Memo to L.Larson, Center for Lake Superior Environmental Studies dated August 31 : 24 p.".

California EPA; Sampling for pesticide residues in California well water. 1995 Update of The Well Inventory Database in California. CA Environmental Protection Agency, Dept. of Pesticide Regulation, 10th Annual Report. (December 1995)

Casely, J.C. The loss of three chloronitrobenzene fungicides from the soil. 1968. Bull. Environ. Contam. Toxicol. 3:180-193.

Cover Jr., J. F., & Boyer, D. M. 1988. Captive reproduction of the San Francisco garter snake, *Thamnophis sirtalis tetrataenia*. *Herpetol. Rev., 19*, 29-33.

Daly, G.L., Lei, Y.D., Teixeira, C., Muir, D.C.G. and F. Wania. 2007. Pesticides in Western Canadian Mountain Air and Soil. Environmental Science and Technology, 41: 6020-6025.

DeWolf, W., J.H.M. Debrujin, W. Selnen and J.L.M. Hermens. Environ. Sci. Technol. 26:1197-1201 (1992).

Fellers, G. M., McConnell, L. L., Pratt, D., & Datta, S. 2004. Pesticides in Mountain Yellow-Legged Frogs (*Rana Mucosa*) from the Sierra Nevada Mountains of California. *Environmental Toxicology and Chemistry, 23*(9), 2170-2177.

Först, C., K. Shafer, A. Andl and L. Steiglitz. Investigation of sorption of some chlorinated pollutants on soil in oil contaminated systems by static and dynamic methods Chemosphere 29:2157-62 (1994).

Hawthorne, S.B., D.J. Miller, P.K.K. Louie, R.D. Butler, and G.G. Mayer. J. Environ. Qual. 25:594-600 (1996).

Health Canada. 2009. Consultation Document on Quintozene - Proposed Re-evaluation Decision - PRVD2009-02. (http://www.hc-sc.gc.ca/cps-spc/pest/part/consultations/_prvd2009-02/quintozene-eng.php).

Hurt, R. 2000. The Elusive California Tiger Salamander. Tideline. 20 (2):1-3. Available online at: http://www.fws.gov/desfbay/Archives/CTSalam/CaTigSal.htm (Accessed on October 3, 2007).

Jordan, T. E., Cornwell, J. C., Walter, R. B., and Anderson, J. T. 2008. Changes in phosphorus biogeochemistry along an estuarine salinity gradient. *Limnology and Oceanography 53*(1), 172-184.

Kanazawa, J. Pestic. Sci. 12:417-424 (1981).

Khan, F., Prakash D., and RK Jain. Development of an HPLC method for determination of pentachloronitrobenzene, hexachlorobenzene and their possible metabolites. BMC Chemical Biology 2011, 11:2 doi:10.1186/1472-6769-11-2

King, R. B. 2002. Predicted and observed maximum prey size - snake size allometry. *Functional Ecology, 16*, 766-772.

Ko, W.H. and J.D. Farley. Phytopathology 59:64-67 (1969).
Korte, F., D. Freitag, H. Geyer, W. Klein, A. G. Kraus and E. Lahaniatis. 1978. Ecotoxicologic profile analysis : A concept for establishing ecotoxicologic priority lists for chemicals. Chemosphere 1:79-102.
Landrum P.F.;Steevens J.A.;Gossiaux D.C.;McElroy M.;Robinson S.;Begnoche L.;Chernyak S.;Hickey J. 2004. Time-Dependent Lethal Body Residues for the Toxicity of Pentachlorobenzene to Hyalella azteca. Environ. Toxicol.
LeNoir, J. S., McConnell, L. L., Fellers, G. M., Cahill, T. M., & Seiber, J. N. 1999. Summertime Transport of Current-use pesticides from California's Central Valley to the Sierra Nevada Mountain Range, USA. *Environmental Toxicology and Chemistry, 18*(12), 2715-2722.
McConnell, L. L., LeNoir, J. S., Datta, S., & Seiber, J. N. 1998. Wet deposition of current-use pesticides in the Sierra Nevada mountain range, California, USA. *Environmental Toxicology and Chemistry, 17*(10), 1908-1916.
Means, J. C. 1995. Influence of salinity upon sediment-water partitioning of aromatic hydrocarbons. *Marine Chemistry, 51*(1), 3-16.
Metcalfe T.L.;Dillon P.J.;Metcalfe C.D. 2008. "Effects of Formulations of the Fungicide, Pentachloronitrobenzene on Early Life Stage Development of the Japanese Medaka (Oryzias latipes)". Chemosphere 71(10): 1957-1962.
Niimi, A.J., H.B. Lee and G.P. Kissoon. Environ. Toxicol. Chem. 8:817-823 (1989).
Olsen, A.R., Snyder, B.D., Stahl, L.L. and J.L. Pitt. 2009. Survey design for lakes and reservoirs in the United States to assess contaminants in fish tissue. Environ Monit Assess, 150:91-100.
Roark J.H., Dale J.L. 1979. The Effect of Turf Fungicides on Earthworms. Ark.Acad.Sci.Proc. 33: 71-79.
Roghair C.J.;Buijze A.;Yedema E.S.E.;Hermens J.L.M. 1994. A QSAR for Base-Line Toxicity to the Midge *Chironomus riparius*. Chemosphere 28(5): 989-997.
Schauerte, W., J.P. Lay, W. Klein and F. Korte. 1982. Long-term fate of organochlorine xenobiotics in aquatic ecosystems. Ecotoxicol. and Environ. Safety 6:560-569.
Scheringer, M., MacLeod, M. and F. Wegmann. 2006. Analysis of four current POP candidates with the OECD Pov and LRTP screening Tool. Available online at: http://www.sust-chem.ethz.ch/docs/POP_Candidates_OECD_Tool.pdf.
Sparling, D. W., Fellers, G. M., & McConnell, L. L. 2001. Pesticides and amphibian population declines in California, USA. *Environmental Toxicology and Chemistry, 20*(7), 1591-1595.
Scheringer, M., MacLeod, M. and F. Wegmann. 2006. Analysis of four current POP candidates with the OECD Pov and LRTP screening Tool. Available online at: http://www.sust-chem.ethz.ch/docs/POP_Candidates_OECD_Tool.pdf.
Stahl, L.L., Snyder, B.D., Olsen, A.R., and J.L.Pitt. 2009. Contaminants in fish tissue from US lakes and reservoirs: a national probabilistic study. Environ Monit Assess, 150:3-19.
Susarla, S., S. Masunaga and Y. Yonezawa. 1996. Transformation of chloronitrobenzenes in anaerobic sediment. Chemosphere 32:967-977.
Swarzenski, P. W., Porcelli, D., Andersson, P. S., & Smoak, J. M. 2003. The behavior of U- and Th-series nuclides in the estuarine environment. Reviews in Mineralogy and Geochemistry *REviews in Mineralogy and Geochemistry, 52*(1), 577-606.
Thompson, T.S., R.G. Treble, D.T. Waite, and A.J. Cessna. 1997. Bull. Environ. Contam. Toxicol. 58: 939-944.

Trenham, P. C., Shaffer, H. B., Koenig, W. D., & Stromberg, M. R. 2000. Life history and demographic variation in the California Tiger Salamander (*Ambystoma californiense*). *Copeia, 2*, 365-377.

USEPA. 1992a. Pesticides in groundwater database. A compilation of monitoring studies: 1971-1991. National summary. USEPA/OPP/OPPTS. USEPA-734-12-92-001.

USEPA. 1992b. National study of chemical residues in fish. United States Environmental Protection Agency. Office of Science and Technology. Volumes 1 and 2. EPA 823-R-92-008 a and b.

USEPA. 1993. *Wildlife Exposure Handbook*. Office of Research and Development, United States Environmental Protection Agency. Available at http://www.epa.gov/ncea/pdfs/toc2-37.pdf (Accessed June 19, 2009).

USEPA. 2003a. Draft chapter "2. Physical and Chemical Properties and Fate" in Volume 2 of NCEA's report "Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds National Academy Sciences (NAS) Review Draft."

USEPA. 2003b. analysis of laboratory and field studies of reproductive toxicity in birds exposed to dioxin-like compounds for use in ecological risk assessment. United States Environmental Protection Agency. Office of Research and Development. National Center for Environmental Assessment. Cincinnati, OH. EPA/600/R-03/114F. April, 2003.

USEPA. 2004. *Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs*. United States Environmental Protection Agency (USEPA). Environmental Fate and Effects Division. Office of Pesticide Programs. Available at http://www.epa.gov/espp/consultation/ecorisk-overview.pdf (Accessed June 19, 2009).

USEPA. 2005. Environmental Fate and Ecological Risk Assessment for the Re-registration of Pentachloronitrobenzene. Environmental Fate and Effects Division, Office of Pesticide Programs, United States Environmental Protection Agency. Feb. 2005.

USEPA. 2006. Reregistration Eligibility Decision (RED) Document for PCNB, EPA-HQ-OPP-2006-0618, July 2006.

USEPA. 2008. Framework for application of the toxicity equivalence methodology for polychlorinated dioxins, furans, and biphenyls in ecological risk assessment. United States Environmental Protection Agency. Office of the Science Advisor, Risk Assessment Forum. EPA 100/R-08/004. June 2008.

USEPA. 2010. 2008 Science Advisory Panel Meeting Follow Up: Assessment of the Bioaccumulation and Long-Range Transport Potential (LRTP) and of Pentachloronitrobenzene (PCNB) and Associated Ecological Risks. Environmental Fate and Effects Division, Office of Pesticide Programs, United States Environmental Protection Agency. March 2010.

USEPA. 2011. Evaluation Guidelines for Ecological Toxicity Data in the Open Literature. Memo from Donald Brady, Director, Environmental Fate and Effects Division, Office of Pesticide Programs, U.S. Environmental Protection Agency. 16 May 2011.

USEPA. 2012. County-Level Usage for Aldicarb, Bensulide, Bifenthrin, Brodifacoum, Chlorothalonil, Diazinon, Difenacoum, Lambda-Cyhalothrin, Methomyl, PCNB, and Pendimethalin in California in Support of a San Francisco Bay Endangered Species Assessment. Memo from Biological and Economic Analysis Division, Office of Pesticide Programs, United States Environmental Protection Agency. Feb. 23, 2012.

USFWS. 2000. Endangered and threatened wildlife and plants: final rule to list the Santa Barbara County distinct population of the California tiger salamander as endangered. Federal Register 65 (184):57242-57263. Available online at: http://frwebgate.access.gpo.gov/cgi-bin/getdoc.cgi?dbname=2000_register&docid=00-24173-filed.pdf (Accessed on October 3, 2007).

USFWS. 2003a. Endangered and threatened wildlife and plants: determination of endangered status for the Sonoma County distinct population segment of the California tiger salamander. Federal Register 68 (53):13498-13520. Available online at: http://ecos.fws.gov/docs/federal_register/fr4072.pdf (Accessed on October 3, 2007).

USFWS. 2003b. Evaluation of the Clean Water Act Section 304(a) Human Health Criterion for Methylmercury: Protection for Threatened and Endangered Wildlife in California. October 2003. Environmental Contaminants Division. Sacramento Fish and Wildlife Office. United States Fish and Wildlife Service. Available at http://www.fws.gov/sacramento/ec/Methylmercury%20Criterion%20Evaluation%20Final%20Report%20October%202003.pdf (Accessed January 25, 2010).

USFWS. 2005. Endangered and threatened wildlife and plants: designation of critical habitat for the Sonoma County distinct population segment of the California tiger salamander. Federal Register 70 (239):74138-74163. Available online at: http://a257.g.akamaitech.net/7/257/2422/01jan20051800/edocket.access.gpo.gov/2005/pdf/05-23701.pdf (Accessed on October 3, 2007).

USFWS/NMFS. 1998. Endangered Species Consultation Handbook: Procedures for Conducting Consultation and Conference Activities Under Section 7 of the Endangered Species Act. Final Draft. United States Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS). Available at http://www.fws.gov/endangered/consultations/s7hndbk/s7hndbk.htm (Accessed June 19, 2009).

USFWS/NMFS/NOAA. 2004. 50 CFR Part 402. Joint Counterpart Endangered Species Act Section 7 Consultation Regulations; Final Rule. *Federal Register* Volume 69. Number 20.Pages 47731-47762. August 5, 2004.

USFWS. 2007. Species Account: Delta Smelt (Hypomesus transpacificus). Sacramento Fish and Wildlife Office. Available at: http://www.fws.gov/sacramento/es/animal_spp_acct/delta_smelt.pdf (Accessed January 6, 2012).

Van den Berg, M., L.S. Birnbaum, M. Denison et al. 2006. The 2005 World Health Organization re-evaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. Toxicol. Sci. 93(2):223-241.

Van Gestel C.A.M., Ma W.C., Smit C.E. 1991. "Development of QSARs in Terrestrial Ecotoxicology: Earthworm Toxicity and Soil Sorption of Chlorophenols, Chlorobenzenes and Dichloroaniline". Sci. Total Environ. 109/110.

Velde, B., & Church, T. 1999. Rapid clay transformations in Delaware salt marshes. *Applied Geochemistry, 14*(5), 559-568.

Wang, X. , S. Harada, M. Watanabe, H. Koshikawa and H. J. Geyer. 1996. Modeling the bioconcentration of hydrophobic organic chemicals in aquatic organisms. Chemosphere 32:1783-93.

Wang, C.H. and F. E. Broadbent. 1972. Kinetics of losses of PCNB and DCNA in three California soils. Soil Sci. Soc. Amer. Proc. 36:742-745.

Willis, GH and McDowell, LL., 1987. Pesticide Persistence on Foliage. Reviews of Environmental Contamination and Toxicology, vol. 100 (1-73).

Wood, T. M., & Baptista, A. M. 1993. A model for diagnostic analysis of estuarine geochemistry. *Water Resources Research 29*(1), 51-71.