Potential Risks of Thiobencarb Use to Federally Threatened California Red-legged Frog (*Rana aurora draytonii*) and Delta Smelt (*Hypomesus transpacificus*)

Pesticide Effects Determinations

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October 19, 2009

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1. Executive Summary

1.1. Purpose of Assessment

The purpose of this assessment is to evaluate potential direct and indirect effects on the California red-legged frog *(Rana aurora draytonii*) (CRLF) and the Delta smelt (*Hypomesus transpacificus*) (DS) arising from Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) regulatory actions regarding use of thiobencarb on rice. In addition, this assessment evaluates whether these actions can be expected to result in effects to designated critical habitat for the CRLF and the DS. This assessment was completed in accordance with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998) and procedures outlined in the Agency's Overview Document (USEPA, 2004).

The CRLF was listed as a threatened species by USFWS in 1996. The species is endemic to California and Baja California (Mexico) and inhabits both coastal and interior mountain ranges. The DS was listed as threatened on March 5, 1993 (58 FR 12854) by the USFWS (USFWS, 2007a). It is mainly found in Suisun Bay and the Sacramento-San Joaquin estuary near San Francisco Bay. During spawning it moves into freshwaters.

1.2. Scope of Assessment

Thiobencarb is a systemic herbicide used to control grasses and broadleaf weeds (USEPA, 1997). It is currently registered as a pre-emergent or early post emergent herbicide for use on dry or wet seeded rice. It is a thiocarbamate class pesticide and its mode of action is inhibition of lipid synthesis (HRAC, 2005). This assessment examines risks of the use of thiobencarb on rice in California to the CRLF and DS. Thiobencarb formulations assessed include emulsifiable concentrates and granular formulations that may be applied via ground or aerial application methods.

Two major degradates were observed in fate studies: 4-chlorobenzoic acid and 4-chlorobenzaldehyde. ECOSAR version 1.0 predicted aquatic toxicity endpoints greater than those predicted and measured for thiobencarb for these degradates, see Appendix B. Additionally, these degradates were not considered to be of toxicological concern in the human health risk assessment completed for the Reregistration Eligibility Decision (RED) and were not recommended to be a human health concern by the Residues of Concern Knowledgebase Subcommittee (ROCKS) (Eckel, 2008; Lewis, 1997; Scollon, 2008). The presence of these degradates and degradates found at less than 10% applied parent equivalents is not expected to alter risk conclusions that are based on the fate, transport, and toxicity of the parent compound alone (see Section 2.2.1 for a complete discussion). Therefore, this assessment estimated exposure to the parent compound alone.

1.3. Assessment Procedures

This assessment was completed in accordance with the USFWS and NMFS *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998) and is consistent with procedures and methodology outlined in the Agency's Overview Document (USEPA, 2004).

1.3.1. 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 CRLF and the DS. Primary constituent elements (PCEs) were used to evaluate whether thiobencarb has the potential to modify designated critical habitat. The Agency evaluated registrant-submitted studies and data from the open literature to characterize thiobencarb toxicity. The most sensitive toxicity value available from acceptable or supplemental studies for each taxon relevant for estimating potential risks to the CRLF and DS and/or their designated critical habitat was used.

1.3.2. Exposure Assessment

1.3.2.a. Aquatic Exposures

Thiobencarb is stable to hydrolysis (MRID 41609012) and aerobic aquatic metabolism and anaerobic aquatic metabolism (MRID 43252001, 42015301). It degrades slowly via photolysis in water (half-life = 190 days) and photolysis in soil (half-life = 420 day dark corrected half-life (MRID 422257801 and 41215312). Thiobencarb dissipates in the environment by binding to soil (K_{OC}s range from 384 – 1435 L/kg, MRID 41215313), by aerobic soil metabolism at the soil/H_2O interface (half-lives ranged from 27-58 days, MRID 43300401, 00040925), and by aqueous photolysis in the presence of photosensitizers (half-life was 12 days, MRID 42257801 and 41215312).

The Tier I Rice Model and a modified version of the Rice Model that accounts for dissipation were used to estimate conservative exposures of thiobencarb in aquatic habitats resulting from runoff and spray drift from different uses. Monitoring data were used to characterize chronic risk to the DS as DS will not be present in rice paddies. Concentrations from aquatic dissipation and aquatic monitoring in California rice growing areas were also used to characterize risk. The peak model-estimated environmental concentration in the rice paddy was 2018 µg/L using the Tier I Rice Model and the 14-day value was 350 µg/L. The 14-day value was estimated using a modified Tier I Rice Model to allow for 14 days of dissipation that would occur in the required water holding period (see Section 3.1). The maximum reported monitoring value in California from surface water data evaluated in this assessment was 170 µg/L before the 14 day holding period was established and 37.4 µg/L after the holding period was established (see Section 3.1.1) (Miller, 1997; Orlando and Kuivila, 2004; Program, 2007). Modeling output showed peak concentrations that are within a reasonable margin of error to the highest peak monitoring data (350-2018 µg/L compared to 170 µg/L) and about 9 - 54 times higher than the peak monitoring data from samples collected after the 14-day holding period was put in place (350-2018 µg/L compared to 37.4 µg/L).

Potential transport mechanisms include pesticide spray drift and secondary drift of volatilized or soil-bound residues leading to deposition onto nearby or more distant ecosystems. Surface water runoff and spray drift are expected to be the major routes of exposure for thiobencarb.

1.3.2.b. Terrestrial Exposures

The T-REX model was used to estimate potential thiobencarb exposures to terrestrial species including birds (surrogate species for terrestrial phase CRLFs), mammals (CRLF prey), and invertebrates (CRLF prey). The AgDRIFT model was used to estimate deposition of thiobencarb on terrestrial and aquatic habitats from spray drift and to determine the distance from thiobencarb use sites that the CRLF and the DS may be at risk of direct or indirect effects. The TerrPlant model was used to estimate thiobencarb exposures to terrestrial-phase CRLF habitat, including plants inhabiting semi-aquatic and dry areas, resulting from uses involving flowable and granular applications. The T-HERPS model was used to allow for further characterization of the dietary exposures of terrestrial-phase CRLFs relative to birds, which were used as a surrogate species for the CRLF.

1.3.3. Toxicity Assessment

Section 4 summarizes the ecotoxicity data available on thiobencarb. Thiobencarb and thiobencarb formulations are moderately to highly toxic to freshwater fish and freshwater invertebrates on an acute exposure basis. The no observed adverse effect concentration (NOAEC) for chronic effects to the striped bass is 21 µg/L, with a lowest observed adverse affect concentration (LOAEC) of 23 µg/L based on posthatch survival (Fujimura *et al.*, 1991). Available chronic toxicity data for aquatic invertebrates include a NOAEC of 1.0 µg/L, with a LOAEC of 3.0 µg/L based on reduction in offspring produced (MRID 00079098). Thiobencarb is slightly to practically nontoxic to birds on an acute oral and subacute dietary exposure basis, and slightly toxic to mammals on an acute oral exposure basis. Thiobencarb is classified as practically nontoxic to honey bees on an acute contact exposure basis. In a reproductive study with the mallard duck a statistically significant decreased number of eggs laid and hatchlings per live three week embryo was observed at concentrations of 300 mg a.i./kg-diet (MRID 00025778). The associated NOAEC was 100 mg a.i./kg-diet. A two generation study on rats with oral exposure to thiobencarb resulted in a NOAEC of less than 20 mg/kg/day with effects on body weight and feeding efficiency observed at 100 mg/kg/day (MRID 40446201). A reproductive NOAEL could not be determined because no reproductive effects were observed at the highest level tested of 100 mg/kg/day. The 96-hr EC_{50} for algae exposed to thiobencarb is 17 µg/L (41690901). The 14-day EC_{50} for duckweed, a vascular plant, was 770 µg/L (MRID 41690901).

1.3.4. Measures of Risk

Acute and chronic risk quotients (RQs) are compared to the Agency's Levels of Concern (LOCs) to identify instances where thiobencarb use has the potential to adversely affect the CRLF or DS 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 thiobencarb 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. Thiobencarb Uses Assessed

In the U.S. and California, thiobencarb is currently registered for use on rice. Only the end-use products approved for use in California are assessed in this document. None of the products registered in California have more than one active ingredient (a.i.) in the formulation; however, several products recommend the use in combination with propanil for control of specific weeds. None of the thiobencarb end-use products are labeled as Restricted Use Pesticides (RUPs).

Based on California Department of Pesticide Regulation Pesticide Use Reporting (CDPR PUR) data, a total of 308,491 to 1,006,327 pounds of thiobencarb were applied annually to rice in California between 1999 and 2006 (Figure 2-2). Sixty-eight percent of the average annual pounds applied were applied in three counties: Colusa, Sutter, and Butte. Colusa and Butte have some CRLF critical habitat. There is no DS critical habitat in these three counties. Average and maximum application rates indicate that thiobencarb is commonly used at the maximum application rate with the average application rate across counties ranging from 3.1 - 4 lbs ai/acre (Table 2-6). The maximum application rate is 4.005 lbs a.i./acre (Table 2-5).

Thiobencarb is currently registered for pre-emergent and early post-emergent control of barnyard grass, junglerice, sprangletop, crabgrass, fall panicum, dayflower, eclipta, False pimpernel, emerged Watergrass, and other weeds. The formulations currently registered include the technical grade which is used to manufacture end-use products, emulsifiable concentrates, and granular formulations. Thiobencarb may be applied via ground and aerial applications. Application methods include broadcast spray, granular applicator, high pressure sprayer, and dilute high volume spray. The timing of application is at dry seeding or wet seeding, prior to rice and weed emergence, or early post emergence of weeds.

1.5. Summary of Conclusions

Based on the best available information, the Agency makes a May Affect, and Likely to Adversely Affect (LAA) determination for the CRLF and the DS from the labeled uses of thiobencarb as described in Table 1-1. The effects determination is based on potential direct and indirect effects to terrestrial-phase CRLF and potential direct and indirect effects to aquatic-phase CRLF and the DS. The LAA determination applies to all currently registered thiobencarb uses in California, *e.g.*, use of thiobencarb on rice.

Additionally, the Agency has determined that there is the potential for habitat modification of designated critical habitat of CRLF and DS from the use of the thiobencarb on rice. A summary of the risk conclusions and effects determinations for each listed species assessed and their designated critical habitat is presented in Table 1-1 and Table 1-2. Further information on the results of the effects determination is included as part of the Risk Description in Section 5.2. Given the LAA determination for the CRLF and the DS and potential effects to designated critical habitat for both species, a description of the baseline status and cumulative effects for the CRLF is provided in Attachment 2 and the baseline status and cumulative effects for the DS is provided in Attachment 4.

Table 1-1. Effects Determination Summary for Potential Effects to the CRLF and DS from the Use of Thiobencarb on Rice in California

Species	Effects Determination	Basis for Determination
California red-legged frog (*Rana aurora draytonii*)	May affect, likely to adversely affect	**Potential for Direct Effects** ***Aquatic-phase (Eggs, Larvae, and Adults)***: Acute RQs for freshwater fish (used as a surrogate for aquatic-phase amphibians) exceed the Agency's LOCs for use of thiobencarb on rice. At the highest RQ (4.59) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.0. Chronic RQs for freshwater fish ranged from 0.26 – 46.10 and exceed the LOC of 1.0. The critical habitat and cultivated crop land cover class overlap. This indicates that direct effects to aquatic-phase CRLF have the potential to occur. ***Terrestrial-phase (Juveniles and Adults)***: The risk of direct adverse effects to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low; however, risk may not be precluded because estimated exposure exceeds the highest doses tested where no mortality occurred for terrestrial birds (the surrogate for terrestrial-phase CRLF) consuming small insects and small mammals. The RQs for chronic risk to terrestrial birds exceed the LOC of 1.0 for birds consuming broadleaf plants/small insects and small mammals. Therefore, chronic risk to the CRLF has the potential to occur. **Potential for Indirect Effects** ***Aquatic prey items, aquatic habitat, cover and/or primary productivity*** Risk quotients for FW fish, FW invertebrates, and aquatic plants exceeded LOCs. For FW invertebrates, the probability of an effect would be approximately 1 in 1.0 (based on the highest RQ of 19.84 and slope of 4.5). Chronic FW invertebrate RQs also exceed the LOC of 1.0. RQs for non-vascular aquatic plants exceed the LOC of 1.0 using modeled and monitoring results. RQs for vascular aquatic plants exceed the LOC of 1.0 based on modeling data in the rice paddy. This indicates that indirect effects to CRLF have the potential to occur due to loss of prey or habitat. RQs for terrestrial plants exceed the LOC of 1.0, indicating that effects to riparian vegetation have the potential to occur. ***Terrestrial prey items, riparian habitat*** CRLFs could be affected as a result of potential impacts to grassy/herbaceous vegetation and reduction of prey items such as small mammals or terrestrial invertebrates. RQs for mammals consuming short grass, tall grass, broadleaf plants, and small insects exceed the acute LOC of 0.1 and chronic LOC of 1.0. The probability of individual effects for mammals is 1 in 2.73 (based on the highest RQ of 0.40 and slope of 4.5). The risk of indirect effects to the CRLF due to a reduction in terrestrial invertebrate prey items is low. Risk may not be precluded for terrestrial invertebrates because the ratio of the EEC to the dose where 15% mortality occurred exceeds the LOC of 0.05. Fifteen percent mortality occurred at the highest dose tested. It is uncertain whether the EC_{50} would result in LOC exceedances for terrestrial invertebrates. RQs for terrestrial plants exceed the corresponding LOC of 1.0.
Delta Smelt	May affect,	**Potential for Direct Effects**

Species	Effects Determination	Basis for Determination
(*Hypomesus transpacificus*)	likely to adversely affect	RQs for freshwater and E/M fish exceed the Agency's LOCs for use of thiobencarb on rice. At the highest RQ (2.82) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.02. Chronic RQs for freshwater and estuarine/marine fish ranged from 0.26 – 13.33 and exceed the LOC of 1.0 when based on aquatic dissipation studies. Critical habitat and the cultivated crop NLCD land cover class overlap. This indicates that direct effects to DS have the potential to occur with the use of thiobencarb on rice.
		Potential for Indirect Effects
		Use of thiobencarb on rice has the potential to adversely affect the DS by reducing available food (aquatic plants and FW and E/M invertebrates), by impacting the riparian habitat of grassy and herbaceous riparian areas, and/or by impacting water quality via effects to aquatic vegetation. Acute and chronic RQs for FW and E/M invertebrates exceed corresponding LOCs indicating that reduction in prey items may occur. For FW invertebrates, the probability of an effect would be approximately 1 in 1.0 (based on the highest RQ of 19.84 and slope of 4.5). For E/M invertebrates the probability of an individual effect is approximately 1 in 1.00 (based on the highest RQ of 3.84 and a slope of 4.5). Chronic RQs for both E/M and FW invertebrates exceed the LOC of 1.0. Some RQs for aquatic plants exceed the LOC of 1.0 indicating that effects on DS habitat and reduction in food may occur. RQs for terrestrial plants exceed the LOC of 1.0 indicating that effects to riparian vegetation have the potential to occur.

Abbreviations: FW = freshwater, E/M = estuarine/marine, CRLF = California Red Legged Frog, DS=delta smelt, RQ=risk quotient

Table 1-2. Effects Determination Summary for Thiobencarb Use on Rice and CRLF and DS Critical Habitat Impact Analysis.

Assessment Endpoint	Effects Determination	Basis for Determination
Modification of aquatic-phase PCEs (DS and CRLF)	Habitat Modification	As described in Table 1-1, the effects determination for the potential for thiobencarb to affect aquatic-phase CRLFs and the DS is LAA. These determinations are based on the potential for thiobencarb to indirectly affect the DS and aquatic-phase CRLF. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF and DS. Therefore, potential effects to aquatic plants and terrestrial (riparian) plants identified in this assessment could result in aquatic habitat modification.
Modification of terrestrial-phase PCE (CRLF)		As described in Table 1-1, the effects determination for the potential for thiobencarb to affect terrestrial-phase CRLFs is LAA. This determination is based on the potential for thiobencarb to directly affect terrestrial-phase CRLFs and their food supply and habitat. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF. Therefore, these potential effects could result in modification of critical habitat.

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 for the CRLF and DS. 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 CRLF and DS 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 USFWS and NMFS (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 California red-legged frog (CRLF, *Rana aurora draytonii*) and delta smelt (DS, *Hypomesus transpacificus*) arising from FIFRA regulatory actions regarding use of thiobencarb on rice. In addition, this assessment evaluates whether use on rice is expected to result in modification of designated critical habitat for the CRLF and/or the DS. This ecological risk assessment has been prepared consistent with the settlement agreements in two court cases. The first case referring to the CRLF is the *Center for Biological Diversity (CBD) vs. EPA et al*. (Case No. 02-1580-JSW(JL)) settlement entered in Federal District Court for the Northern District of California on October 20, 2006. This assessment also addresses the DS for which thiobencarb was alleged to be of concern in a separate suit (*Center for Biological Diversity (CBD) vs. EPA et al*. (Case No. 07-2794-JCS)).

In this assessment, direct and indirect effects to the CRLF and DS and potential modification to designated critical habitat for the CRLF and DS are evaluated in accordance with the methods described in the Agency's Overview Document (USEPA, 2004). The effects determinations for each listed species assessed is based on a weight-of-evidence method that relies heavily on an evaluation of risks to each taxon relevant to assess both direct and indirect effects to the listed species and the potential for modification of their designated critical habitat (*i.e*., a taxon-level approach). Screening level methods include use of standard models such as Tier I Rice Model, T-REX, TerrPlant, and AgDRIFT, all of which are described at length in the Overview Document. In addition, T-HERPS has been used to refine estimates of exposure and risk to amphibians. Use of such information is consistent with the methodology described in the Overview Document (USEPA, 2004), which specifies that "the assessment process may, on a case-by-case basis, incorporate additional methods, models, and lines of evidence that EPA finds technically appropriate for risk management objectives" (Section V, page 31 of USEPA, 2004).

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 thiobencarb 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 thiobencarb 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 CRLF and DS 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 thiobencarb in accordance with current labels:

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

The CRLF and the DS have designated critical habitats associated with them. Designated critical habitat identifies specific areas that have the physical and biological features, (known as primary constituent elements or PCEs) essential to the conservation of the listed species. The PCEs for CRLFs are aquatic and upland areas where suitable breeding and non-breeding aquatic habitat is located, interspersed with upland foraging and dispersal habitat. PCEs for the DS include characteristics required to maintain habitat for spawning, larval and juvenile transport, rearing, and adult migration.

If the results of initial screening-level assessment methods show no direct or indirect effects (no LOC exceedances) upon individuals or upon the PCEs of the species’ designated critical habitat, a “no effect” determination is made for use of thiobencarb as it relates to each species and its designated critical habitat. If, however, potential direct or indirect effects to individuals of each species are anticipated or effects may impact the PCEs of the designated critical habitat, a preliminary “may affect” determination is made for the FIFRA regulatory action regarding thiobencarb.

If a determination is made that use of thiobencarb “may affect” a listed species or its designated critical habitat, additional information is considered to refine the potential for exposure and for effects to each species and other taxonomic groups upon which these species depend (*e.g.*, prey items). Additional information, including spatial analysis (to determine the geographical proximity of the assessed species’ habitat and thiobencarb use sites) and further evaluation of the potential impact of thiobencarb on the PCEs is also used to determine whether modification of designated critical habitat may occur. Based on the refined information, the Agency uses the best available information to distinguish those actions that “may affect, but are not likely to adversely affect” from those actions that “may affect and are likely to adversely affect” the assessed listed species and/or result in “no effect” or potential modification to the PCEs of its designated critical habitat. This information is presented as part of the Risk Characterization in Section 5 of this document.

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 thiobencarb is expected to directly impact living organisms within the action area (defined in Section 2.7), critical habitat analysis for thiobencarb 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 (*i.e*., the biological resource requirements for the listed species associated with the critical habitat or important physical aspects of the habitat that may be reasonably influenced through biological processes). Activities that may modify critical habitat are those that alter the PCEs and appreciably diminish the value of the habitat. Evaluation of actions related to use of thiobencarb that may alter the PCEs of the assessed species’ critical habitat form the basis of the critical habitat impact analysis. Actions that may affect the assessed species’ designated critical habitat have been identified by the Services and are discussed further in Section 2.6.

2.2. Scope

Thiobencarb is a systemic herbicide used to control grasses and broadleaf weeds (USEPA, 1997). It is currently used to control weeds in rice paddies. It may be applied as a pre-emergent or early post emergent herbicide to dry or wet seeded rice. It is a thiocarbamate class pesticide and its mode of action is inhibition of lipid synthesis (HRAC, 2005).

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 thiobencarb in accordance with the approved product labels for California is "the action" relevant to this ecological risk assessment.

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

2.2.1. Evaluation of Degradates

This ecological risk assessment includes all potential ecological stressors resulting from the use of thiobencarb, including thiobencarb and its potential degradates of concern. Degradates of concern may include those that are found at significant concentrations (>10% by weight relative to parent) in available degradation studies and/or those that are of toxicological concern. The only major degradates (defined as those representing 10% or more of the applied radiation of the parent test substance) identified were 4-chlorobenzoic acid (56% of applied radiation at 30 days) and 4-chlorobenzaldehyde (29.4% at 14 days), both in a sensitized aquatic photodegradation study. Both of these degradates are expected to be soluble and mobile in water. Both should be subject to further degradation by metabolism (based on their simple structures), but the study was terminated at 30 days.

Neither of these degradates was analyzed for in the aquatic field dissipation studies, thus we do not know if their formation in laboratory studies means that they will form in the field. The only degradates measured in the aquatic field studies were thiobencarb sulfoxide and 4-chlorobenzylmethylsulfone) which is a possible precursor to 4-chlorobenzoic acid and 4-chlorobenzaldehyde.

A summary of formation pathways of the two major degradates, 4-chlorobenzoic acid and 4-chlorobenzaldehyde, is presented in Table 2-1. Appendix A provides additional information on the percentage of applied parent equivalents in each environmental fate study.

Table 2-1. Degradate Occurrence Summary for Thiobencarb

Study **MRID**	**4-chlorobenzoic acid** **CAS Number:** **74-11-3**	**4-chlorobenzaldehyde** **CAS Number:** **104-88-1**
Hydrolysis 41609012		
Aqueous Photolysis 42257801	3.9% @ 30 days max/last	1.8% @ 21 days max 1.8% @ 30 days last
Aqueous Photolysis 42257801 (sensitized)	**56**% @ 30 days max/last	**29.4**% @ 14 days max 3.7% @ 30 days last
Soil Photolysis 41215312	1% @ day 9 max 1.1% @ day 21 last	
Aerobic Soil Metabolism 43300401	0.6% @ 120 days max 0.13% @ 366 days last	
Aerobic Soil metabolism 00040925	2.6% @ 7 days max	
Aerobic Aquatic Metabolism 42015301	0.5% @ 7 days max 0.2% @ 30 days last	
Anaerobic Aquatic Metabolism 43252001	2.2% @ 363 days max/last	
Aquatic field Dissipation/wet seeded 43404005	Not analyzed	Not analyzed
Aquatic Field Dissipation/dry seeded 42003404	Not analyzed	Not analyzed

No ecological toxicity data were found in the ECOTOX database for 4-chlorobenzoic acid or 4-chlorobenzaldehyde; however, ECOSAR version 1.0 predicted aquatic toxicity endpoints greater than those predicted and measured for thiobencarb, see Appendix B. These degradates were not considered to be of toxicological concern in the human health risk assessment completed for the Reregistration Eligibility Decision (RED) and were not recommended to be treated as a human health concern by the ROCKS Committee (Eckel, 2008; Lewis, 1997). The toxicity of 4-chlorobenzoic acid and 4-chlorobenzaldehyde was assumed to be less than that of thiobencarb (see Appendix B). Concentrations of the other degradates were a very small percentage (<8.3%) of the amount of thiobencarb applied or were only present in a few fate studies, suggesting that they would be present at lower concentrations than those estimated for thiobencarb. The estimated toxicity based on structure activity relationships (ECOSAR version 1.0) or the similarity of the structure to thiobencarb indicates that the toxicity of these compounds is similar to or less than that of thiobencarb (see Appendix B). The presence of these degradates is not expected to alter risk conclusions that are based on the fate, transport, and toxicity of the parent compound alone.

2.2.2. Evaluation of Mixtures

The Agency does not routinely include an evaluation of mixtures of active ingredients (either those mixtures of multiple active ingredients in product formulations, or those in the applicator's tank, in its risk assessments. In the case of product formulations of active ingredients (registered products 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 ingredients, 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). Thiobencarb does have one nationally registered end-use product that is co-formulated with propanil; however, this product is not registered for use in California[1] (EPA Registration Number 07108500030). Therefore, none of the thiobencarb products assessed here contains more than one active ingredient. Labels registered in California do recommend use of thiobencarb in tank mixes with propanil and thiobencarb is often mixed or used with propanil (see labels).

The results of available toxicity data for mixtures of thiobencarb with other pesticides are presented in Appendix C. The limited data available do no allow a comparison of the toxicity results of thiobencarb alone versus thiobencarb and propanil or with other chemicals. If chemicals that show synergistic effects with thiobencarb are present in the environment in combination with thiobencarb, the toxicity of thiobencarb may be increased, offset by other environmental factors, or even reduced by the presence of antagonistic contaminants if they are also present in the mixture. The variety of chemical interactions presented in the available data set suggest that the toxic effect of thiobencarb, in combination with other pesticides used in the environment, can be a function of many factors including but not necessarily limited to (1) the exposed species, (2) the co-contaminants in the mixture, (3) the ratio of chemical concentrations, (4) differences in the pattern and duration of exposure among contaminants, and (5) the differential effects of other physical/chemical characteristics of the receiving waters (*e.g.*, organic matter present in sediment and suspended water). Quantitatively predicting the combined effects of all these variables on mixture toxicity to any given taxa with confidence is beyond the current capabilities. However, a qualitative discussion of implications of the available pesticide mixture effects data involving thiobencarb on the confidence of risk assessment conclusions is addressed as part of the uncertainty analysis for this effects determination.

[1] RiceBeaux (EPA Registration Number 41085-30, label date 11/05/2008) is a Section 3 national label and contains propanil; however, it is not for sale or registered for use in California.

2.3. Previous Assessments

2.3.1. Reregistration Eligibility Decision (RED)

Thiobencarb was first registered in 1975 (USEPA, 1994). A Reregistration Eligibility Decision (RED) for thiobencarb was signed in December 1997 (USEPA, 1997). In the RED, the following mitigation measures were required that are especially relevant to the ecological risk assessment.

- Include label warnings preventing application to rice fields with catfish/crayfish farming, and preventing application to rice fields adjacent to catfish or crayfish ponds.
- Where weather conditions permit, it is required that flood waters not be released within 14 days.
- Avoid application of this product within 24 hours of rainfall, or when heavy rain is expected to occur within 24 hours.

Product Reregistration for all thiobencarb products was completed on April 10, 2006 (Ballard and Errico, 2009). Therefore, all measures required by the RED should be reflected on current product labels.

The following data gaps were identified in the EFED science chapter or in follow-up actions to the RED (Mastrota, 1997, Not Specified): chronic toxicity test with shrimp did not have a definitive NOAEC, avian acute dietary toxicity for waterfowl, avian chronic toxicity for a waterfowl species, freshwater fish full life cycle study, seedling emergence on lettuce and ryegrass .

The ecological risks of the various uses are summarized below for California for non-listed species (Mastrota, 1997).

- Use of liquid formulations pose some acute risk to mammals. The acute risk to birds is minimal.
- Use of liquid formulations pose a high chronic risk to birds and mammals. The chronic risk from granular formulations could not be assessed.
- Use of thiobencarb on rice in California poses a risk of causing chronic effects to aquatic organisms in the smaller drains and waterways, but not in the larger rivers.
- Its use poses minimal risk of acute effects to fish and aquatic invertebrates.
- Minimal risk of both acute and chronic effects is expected for all estuarine/marine (E/M) organisms in California.
- Spray drift from aerial application of liquid thiobencarb on rice poses a high risk to nontarget terrestrial and semi-aquatic plants. Drift of granular thiobencarb and spraying of liquid thiobencarb applied with ground equipment pose minimal risk to these plants.
- All uses of thiobencarb on rice may pose a risk of killing emerging seedlings of aquatic plants, especially aquatic grasses. Use of thiobencarb on rice may pose a risk to aquatic algae in smaller drains and waterways in California.

2.3.2. Ecological Risk Assessment for Proposed Use on Wild Rice

An ecological risk assessment was completed on a proposed use of an emulsifiable concentrate formulation for aerial or ground application to soil in wild rice paddies at a rate of 2 to 4 lbs a.i./acre with one application per season (Davy, 2008). In this risk assessment, the Tier I Rice Model was used to estimate exposure. This model was not available when the RED was completed. Many other changes in methodologies of assessing risk have occurred between 1997 and 2008. The following risk concerns were identified in the 2008 assessment:

- acute and chronic risk to aquatic organisms;
- risk to aquatic plants;
- acute risk to mammals;
- chronic risk to birds and mammals;
- risk to terrestrial plants; and
- risk to federally listed birds (and reptiles), mammals, freshwater and E/M fish (and amphibians), freshwater and E/M aquatic invertebrates, and aquatic and terrestrial plants.

2.3.3. Effects Determination for Thiobencarb for Pacific Anadromous Salmonids

A determination was made by the Field and External Affairs Division that thiobencarb use on rice is not likely to adversely affect Federally listed threatened and endangered salmon and steelhead, nor is it likely to adversely modify their designated critical habitat (Turner, 2002). This determination was largely based on, "maximum residues found in natural waters providing habitat for salmon and steelhead have been consistently below levels of concern for acute toxicity to fish or indirectly to their invertebrate food supply". The Office of Pesticide Programs (OPP) requested initiation of Endangered Species Act (ESA) Section 7(a)(2) consultation with the National Marine Fisheries Service in 2002 (Williams, 2002).

2.3.4. Registrant Submitted Data Not Considered In Previous Ecological Risk Assessments

New data are available that were not used in previous risk assessments. These studies include an 850.1075 acute freshwater fish toxicity test (MRID 46091401), freshwater fish full life cycle test (MRID 45695101), 28-day sediment toxicity test for the midge (*Chironomus riparius*) (MRID 46091402), 850.3200 acute contact toxicity to the honey bee (MRID 46059804), and an acute earthworm study (MRID 46059803).

2.4. Stressor Source and Distribution

2.4.1. Environmental Fate Properties

Thiobencarb is generally non-persistent in the water column, due to dissipation to soil, but moderately persistent in soils and sediments. Thiobencarb dissipates in the environment by binding to soil, by aerobic soil metabolism at the soil/H_2O interface, and by aqueous photolysis in the presence of photo-sensitizers. Thiobencarb K_{OC}s range from 384 – 1438 L/kg indicating it is moderately mobile to slightly mobile and it does have the potential to reach groundwater in highly vulnerable areas (based on the FAO Mobility Classification) (FAO, 2000). When used on rice, thiobencarb is more likely to be found in the soil than in the paddy water. Greater quantities of thiobencarb are associated with soil when applied preflood to soil rather than in standing water. The portion of thiobencarb associated with soil was approximately 10 times more when applied pre-flood to soil than when applied to standing water, primarily since thiobencarb has time to bind to soil prior to flooding. As a result, sensitized aqueous photolysis is expected to be more significant as a dissipation route when thiobencarb is applied to water than when it is applied to dry soil, due to a greater amount of thiobencarb remaining in paddy water containing natural photo-sensitizers.

Thiobencarb has a water solubility of 30 ppm, a vapor pressure of 1.476 x 10^{-6} Torr at 20^{o}C (Ahrens, 1994) and 2.2×10^{-5} at 23^{o}C (MRID 00044507), and an estimated Henry's Law Constant of 2.49 x 10^{-7} atm m^3/mol. It is stable to hydrolysis, non-sensitized aqueous photolysis, soil photolysis, anaerobic aquatic metabolism, and aerobic aquatic metabolism. In an aqueous photolysis study with and without the use of photo-sensitizers (acetone), the half-lives were 12 and 190 days, respectively (MRID 42257801). Since some humic substances in natural waters have been shown to act as photo-sensitizers, the 12-day half-life may be more relevant. Thiobencarb also degraded moderately slowly under aerobic conditions with calculated half-lives of 27-58 days in soils that typically support rice production.

Thiobencarb slowly mineralizes in soil without forming significant quantities of non-volatile degradates; however, major degradates are formed with photosensitized photolysis. The major degradate in both the aqueous photolysis and soil metabolism studies was 4-chlorobenzoic acid, reaching 56 and 5 % of applied parent equivalents respectively. Another major degradate observed in photolysis studies was 4-chlorobenzaldehyde at a maximum of 29.4% applied parent equivalents. CO_2 and bound residues are the primary products from soil metabolism studies, occurring in proportions of 42-77 and 23-42 % of applied, respectively. Aqueous residues did not exceed 4.5 % of applied radioactivity in soil metabolism studies.

Parent thiobencarb was moderately mobile to slightly mobile (based on the FAO Mobility classification system) in the tested soils with Freundlich K_{ads} values of 5.42-20 L/kg. The K_{oc} values ranged from 384-1435 L/kg. 4-Chlorobenzoic acid was mobile to moderately mobile (based on the FAO Mobility classification system) in the tested soils with Freundlich K_{ads} values of 0.74-3.26 L/kg. The corresponding K_{oc} values ranged from 84-416 L/kg. Mobility generally decreased with increasing clay content, increasing organic matter content, and increasing cation exchange capacity.

Results from an aquatic field dissipation study in Louisiana, where thiobencarb was applied as a spray directly to soil and flooded seven days later, show half-lives of 5.8 days in flood water and 36 days in hydrosoil or sediment. The median ratio of soil:water thiobencarb residues was 63.5:1.

In two field studies in California rice paddies where granules were applied into standing water, the dissipation half-lives in flood water were 8.7 days (guideline study) and 4.5 days (literature review, Ross and Sava, 1986). The half-lives in hydrosoil or sediment were 153 and 56 days, respectively. The median ratios of soil:water thiobencarb residues were 5.6:1 and 6.6:1.

Thiobencarb moderately accumulated in bluegill sunfish with maximum bioconcentration factors of 128x, 639x, and 411x for edible (muscle) tissue, nonedible tissue, and whole fish, respectively. Depuration is rapid, with 93-95% of the accumulated [^{14}C]residues being eliminated from the tissues in three days. The degradates 4-chlorobenzylmethylsulfoxide, thiobencarb sulfoxide, desethylthiobencarb, and 2-hydroxythiobencarb were identified in edible and nonedible tissue.

Table 2-1 lists the environmental fate properties of thiobencarb, along with the major and minor degradates detected in the submitted environmental fate and transport studies.

Table 2.1 Summary of Physical/Chemical and Environmental Fate Properties of Thiobencarb

Parameter	Value	Source	Comment
Chemical name	S-4-chlorobenzyl diethylthiocarbamate	Ahrens, 1994	
Molecular weight	257.78 g/mole	Ahrens, 1994	
Water Solubility	30 mg/L in water at 25º	Ahrens, 1994	
Vapor pressure	1.476×10^{-6} mm Hg at 20Cº 2.2 x 10-5 mm Hg at 23ºC	Ahrens, 1994 MRID 00044507 as referenced in (Knizner, 1995)	
Henry's Law Constant	2.49×10^{-7} atm-m^3/mol		Calculated from vapor pressure at 23ºCand water solubility at 25ºC
Log K_{ow}	3.42 1.3 – 1.6	Ahrens, 1994 MRID 0044507 as referenced in (Knizner, 1995)	
pK_a	none	Ahrens, 1994	
Hydrolysis half-life	Stable	MRID 41609012	Stable at pH 5, 7, and 9
Aqueous photolysis half-life	190 days 12 days in presence of 1% acetone	MRID 42257801	In non-sensitized, sterile pH7 buffer at 25ºC; stable in dark control
Soil photolysis half-life	168 days (irradiated) 280 days (dark control)	MRID 41215312	Dark-corrected half-life is 420 days
Aerobic soil metabolism half-life	58 days (0-56 day data) Stockton Clay adobe soil (CA) 37 days (Clay soil, Biggs, CA) 27 days (Silty Clay Loam, Crowley, LA)	MRID 43300401 MRID 00040925	
Anaerobic aquatic metabolism half-life	Stable	MRID 43252001	
Aerobic aquatic metabolism half-life	Stable	MRID 42015301	
Soil organic carbon partition coefficient in L/kg (K_{oc})	1084 (sandy loam) 384 (Loam) 618 (silty clay) 1027 (clay loam) 1435 (silt loam)	MRID 41215313	

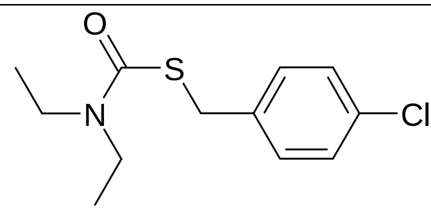
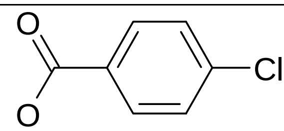
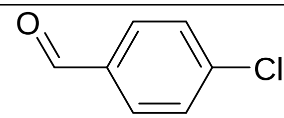
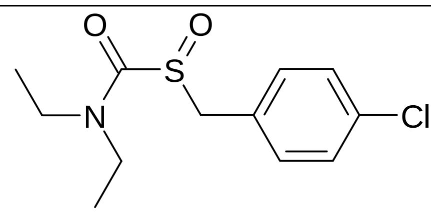
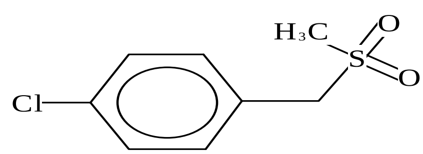
A summary of degradates found in the California and Louisiana aquatic field dissipation studies is found in Table 2-2.

Table 2-2. Maximum Amounts of Thiobencarb and Metabolites Found in Flood Water in Field Dissipation Studies

Study Type	Metabolite (Maximum in Flood Water)	MRID
Louisiana Aquatic Field Dissipation (dry-seeded rice) 4 lb/acre, aerial, spray, flooded to 4.5 inches at 7 days post-application	Parent Thiobencarb **Max. 12.2-14.1 ppb at 3 days post-flood (PF)** 5.6-10.5 ppb at 7 days PF Less than 1 ppb 28-70 days PF Thiobencarb sulfoxide **Max. 16-13.4 ppb at 1 day PF** 2.6-5.2 ppb at 3 days PF 0.8-1.5 ppb at 7 days PF Less than 0.9 ppb at 14 to 70 days PF 4-chlorobenzylmethylsulfone **Max. 4.8-5.8 ppb at 5 days PF** 1.4-1.8 ppb at 14 days PF Less than 0.5 ppb 21-70 days PF	MRID 42003404
California Aquatic Field Dissipation (wet-seeded rice) 4 lb/acre, aerial, granular, flooded to 6 inches at time of application	Parent Thiobencarb 266 ppb at zero days **Max. 438 ppb at 3 days** 1.0 ppb at 92 days Thiobencarb sulfoxide 4.4 ppb at zero days **Max. 22 ppb at 3 days** Non-detectable at 33 days 4-chlorobenzylmethylsulfone 1.11-3.14 ppb at day zero **Max. 6.38-10.0 ppb (avg. 8.3) at day 10** 5.15-6.95 ppb at day 21 Nondetect-1.52 ppb at day 33 Nondetect-1.1 ppb at day 92	MRID 43404005
Ross and Sava, 1986 (wet-seeded) 4 lb/acre, aerial, spray, water depth 10.4 inches	Parent Thiobencarb **Max. 576 ppb at 4 days** No Degradates measured	none

The chemical structures of the parent and some degradates are presented in Table 2-3. More information on degradates is available in Appendix A.

Table 2-3. The Chemical Structure of Thiobencarb and its Metabolites

Thiobencarb	
4-chlorobenzoic acid	
4-chlorobenzaldehyde	
Thiobencarb sulfoxide	
4-chlorobenzylmethylsulfone	

Volatization of Thiobencarb

A small amount of applied thiobencarb may volatilize and be transported in air. The vapor pressure indicates that thiobencarb has an intermediate volatility from dry, non-adsorbing surfaces and the estimated Henry's Law Constant indicates it is non-volatile from water (based on classifications in the terrestrial field dissipation OPPTS Guidline 835.6100). Thiobencarb has been measured in air in rice growing areas of California. The measured flux rate for air is 23 $ng/cm^2/hr$ and 58 $ng/cm^2/hr$ for water (Woodrow *et al.*). Thiobencarb was predicted to potentially travel up to 5 km from the site of application (Woodrow *et al.*). Evaporation percentages of the amount applied are low (0.90 and 0.10% a few days after application to wet seeded rice) (Ceesay, 2002). The atmospheric degradation half-life is 0.421 days (estimated using EPISuite Version 4.0, Appendix M)) inidicating that long range transport is not likely to occur. However, it has been found at trace concentrations in air and precipitation (see Section 3.1.1.f and Section 3.1.1.g)

2.4.2. Environmental Transport Mechanisms

Potential transport mechanisms include rice paddy water discharge, spray drift, and secondary drift of volatilized or soil-bound residues leading to deposition onto nearby or more distant ecosystems. Rice paddy water discharge and spray drift are expected to be the major routes of exposure for thiobencarb.

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). Thiobencarb was not looked for or found in these studies; however, the vapor pressure of thiobencarb indicates that some atmospheric transport may occur (see Section 2.4.1). 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 CLRF are located east of the Central Valley. The magnitude of transport via secondary drift depends on thiobencarb'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 thiobencarb 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 considered in evaluating the potential for atmospheric transport of thiobencarb to locations where it could impact the CRLF and DS.

In general, deposition of drifting or volatilized pesticides is expected to be greatest close to the site of application. Computer models of spray drift (AgDRIFT) are used to determine potential exposures to aquatic and terrestrial organisms via spray drift. The distance of potential impact away from the use sites is determined by the distance required to fall below the LOC. The highest RQ/LOC ratio was observed for mammals (see Section 5.1).

2.4.3. Mechanism of Action

Thiobencarb is a thiocarbamate herbicide. Thiocarbamates inhibit development of seedling shoots and roots as they emerge from the seed (Ware and Whitacre, 2004). The mode of action is inhibition of lipid synthesis but not via ACCase inhibition (HRAC, 2005). Thiobencarb works best when applied after rice seedlings are at the 3-6 leaf stage and when applied prior to weed emergence (Ampong-Nyarko and DeDatta, 1991). Post emergent applications should be applied after the 1.5-leaf stage of rice to obtain the best weed kill without damage to rice (Ampong-Nyarko and DeDatta, 1991).

2.4.4. Use Characterization

Analysis of labeled use information is the critical first step in evaluating the federal action. The current labels for thiobencarb represent the FIFRA regulatory action; therefore, labeled use and application rates specified on the labels 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.

Thiobencarb was first registered under FIFRA in 1975 (USEPA, 1994). There are currently 28 thiobencarb products registered in the U.S. Seven are Section 3 National registrations with one being a technical grade for use in the manufacture of end-use products. Twenty-one Section 24 Special Local Needs products are registered, two of which may be used in California (see Table 2-4). None of the products registered in California have more than one active ingredient in the formulation; however, several products recommend the use in combination with propanil for control of specific weeds.

Thiobencarb is currently registered for use on rice for pre-emergent and early post-emergent control of barnyard grass, junglerice, sprangletop, crabgrass, fall panicum, dayflower, eclipta, False pimpernel, emerged Watergrass, and other weeds. The formulations currently registered include the technical grade which is used to manufacture end-use products, emulsifiable concentrates, and granular formulations. Thiobencarb may be applied via ground and aerial applications. Application methods include broadcast spray, granular applicator, high pressure sprayer, and dilute high volume spray. The timing of application is at dry seeding or wet seeding, prior to rice and weed emergence, or early post-emergence of weeds.

Table 2-4 presents the uses and corresponding application rates and methods of application considered in this assessment. There is no further pending mitigation (*i.e.*, reduction in application rates, cancellation of uses, label language on buffers and spray drift requirements, *etc*.) that may impact the conclusions of this assessment in the near future. The following lists some of the mitigation measures required through the reregistration (RED) process that could impact the evaluation of ecological risk.

- Do not apply this product to rice fields with catfish/crayfish farming.
- Do not apply this product on rice fields adjacent to catfish or crayfish ponds.
- When applying to rice fields, do not release permanent flood water within 14 days of application of this product (where weather permits).

- Avoid application of this product within 24 hours of rainfall, or when heavy rain is expected to occur within 24 hours.

The reregistration process was completed on April 10, 2006 and all measures required by the RED should be reflected on current product labels (Ballard and Errico, 2009). Table 2-5 presents the uses and corresponding application rates and methods of application considered in this assessment. Appendix D includes a summary of the labels prepared by BEAD.

Table 2-4. Summary of Thiobencarb Products Registered under Section 3 and Section 24c Special Local Needs for use In California**

Product Name (EPA Reg. No.)	Registrant	Percent Active Ingredient	Formulation	Use(s)
Bolero Technical (63588-4)	K.-I Chemical U.S.A. Inc.	97.4 thiobencarb	Technical*	Used to make end-use products
Bolero 8 EC (59639-79)	Valent U.S.A. Corporation	84 thiobencarb	Emulsifiable Concentrate	Rice
Valent Bolero 10 G (59639-80)	Valent U.S.A. Corporation	10 thiobencarb	Granular	Rice
Bolero 15 G (59639-112)	Valent U.S.A. Corporation	15 thiobencarb	Granular	Rice
Bolero 8 EC (63588-6)	K.-I Chemical U.S.A. Inc	84 thiobencarb	Emulsifiable Concentrate	Rice
Bolero 15 G (63588-14)	K.-I Chemical U.S.A. Inc	15 thiobencarb	Granular	Rice
Valent Bolero 8 EC (CA930003)	Valent U.S.A. Corporation	84 thiobencarb	Emulsifiable Concentrate	Rice
Valent Bolero 10 G (CA970036)	Valent U.S.A. Corporation	10 thiobencarb	Granular	Rice

*The technical grade chemical is only labeled for use in producing end use products.
** RiceBeaux (EPA Registration Number 41085-30, label date 11/05/2008) is a Section 3 national label; however, it is not for sale or registered for use in California.

Table 2-5. Rice Uses of Thiobencarb Assessed in California

Formulation	Maximum Single Application Rate (lbs a.i./acre)	Maximum Application Rate per Year (lbs a.i./acre)	Maximum Number of Applications per Year	Minimum Retreatment Interval	Appl. Methods
Emulsifiable Concentrate	3 4 4	4 4 NS	NS NS NS	NS NS NS	Aircraft Ground
Granular	4.005	4.005	1	NS	Granule Applicator// Broadcast

According to the United States Geological Survey's (USGS) national pesticide usage data (based on information from 1999 to 2004), an average of 1,057,714 lbs of thiobencarb is applied nationally in the U.S. (Figure 2-1). All of that is applied to rice as that is the only registered use for thiobencarb. The highest usage, geographically, is in Central California, Texas, Louisiana, Mississippi, and Arkansas.

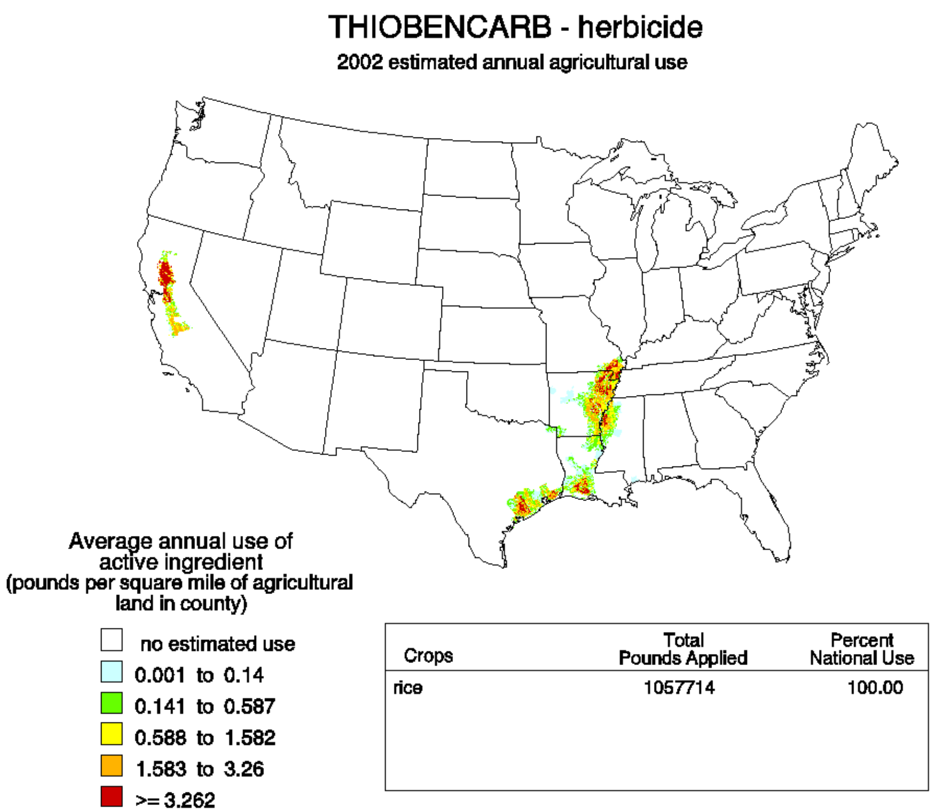


Figure 2-1.Thiobencarb Use in Total Pounds per County

(from http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=02&map=m1903)[2]

The Agency's Biological and Economic Analysis Division (BEAD) provides an analysis of both national- and county-level usage information (Carter and Kaul, 2009) 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 thiobencarb by county in this California-specific assessment were generated using CDPR PUR data. Eight years (1999-2006) 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.[5] 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 application rate across all eight years. The units of area treated are also provided where available.

Some uses reported in the CDPR PUR database are different than those considered in the assessment (alfalfa, cotton, cucumber, wine grape, nursery-outdoor flower, nursery-outdoor plants in containers, research commodity, tomato, and uncultivated agricultural areas). The uses considered in this risk assessment represent all currently registered uses according to a review of all current labels. No other uses are relevant to this assessment. Any other reported use, such as may be seen in the CDPR PUR database, represent either historic uses that have been canceled, mis-reported uses, or mis-use. Historical uses, mis-reported uses, and misuse are not considered part of the federal action and, therefore are not considered in this assessment

According to the CDPR PUR database, a total of 308,491 to 1,006,327 pounds of thiobencarb were applied annually to registered crops in California between 1999 and 2006 (Figure 2-2). The average total annual number of pounds applied by county over that eight year period was 635,896 lbs. Figure 2-3 shows the reported average annual number of pounds used in each county between 1999 and 2006. Sixty-eight percent of the average annual pounds applied were

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[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.

[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.usda.gov/nass/pubs/estindx1.htm#agchem.

[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.

[5] Most pesticide applications to parks, golf courses, cemeteries, rangeland, pastures, and along roadside and railroad rights of way, and postharvest treatments of agricultural commodities are reported in the database. The primary exceptions to the reporting requirement are home-and-garden use and most industrial and institutional uses (http://www.cdpr.ca.gov/docs/pur/purmain.htm).

applied in three counties: Colusa, Sutter, and Butte. Colusa, Sutter, Butte, Glenn, Yuba, Yolo, Placer, and Sacramento used on average greater than 20,000 pounds thiobencarb a year. Average and maximum application rates indicate that thiobencarb is commonly used at the maximum application rate with the average application rate across counties ranging from 3.1 - 4 lbs ai/acre (Table 2-6).

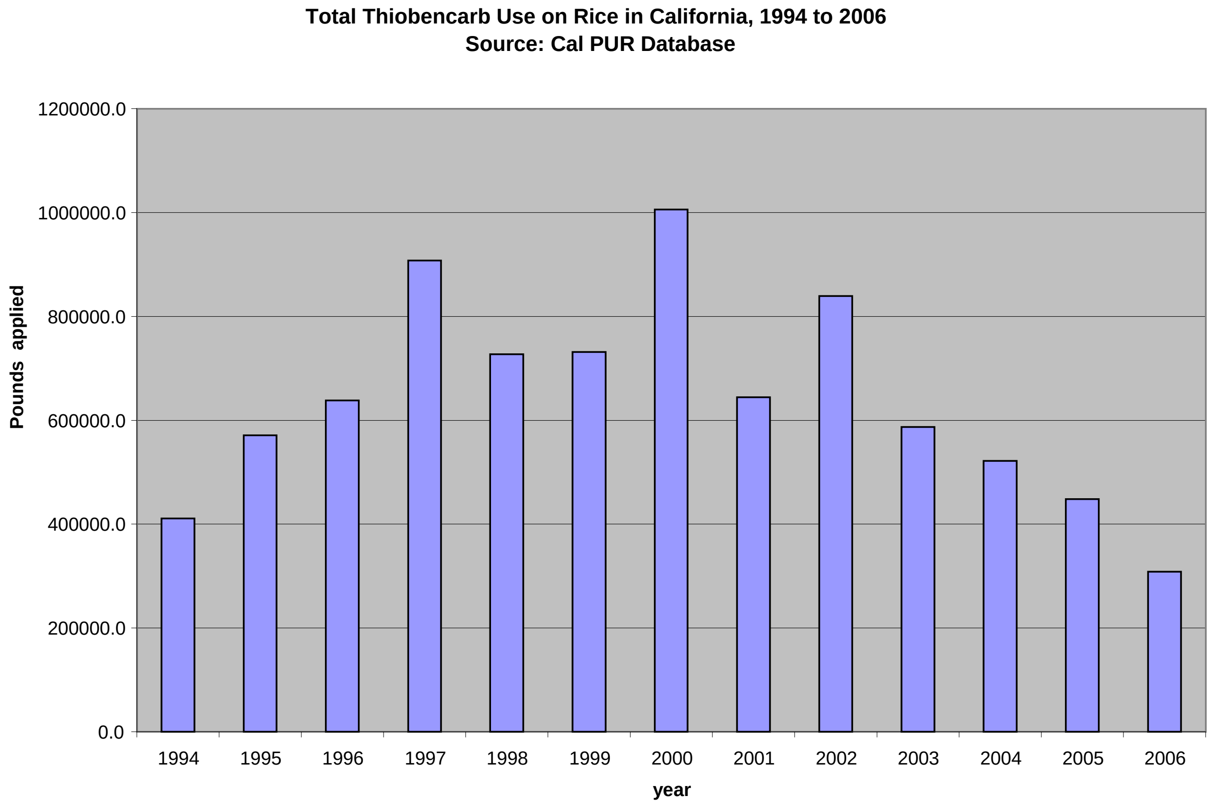

Figure 2-2. Annual Pounds Thiobencarb Applied in California per Year Between 1994 and 2006 (source California PUR database).

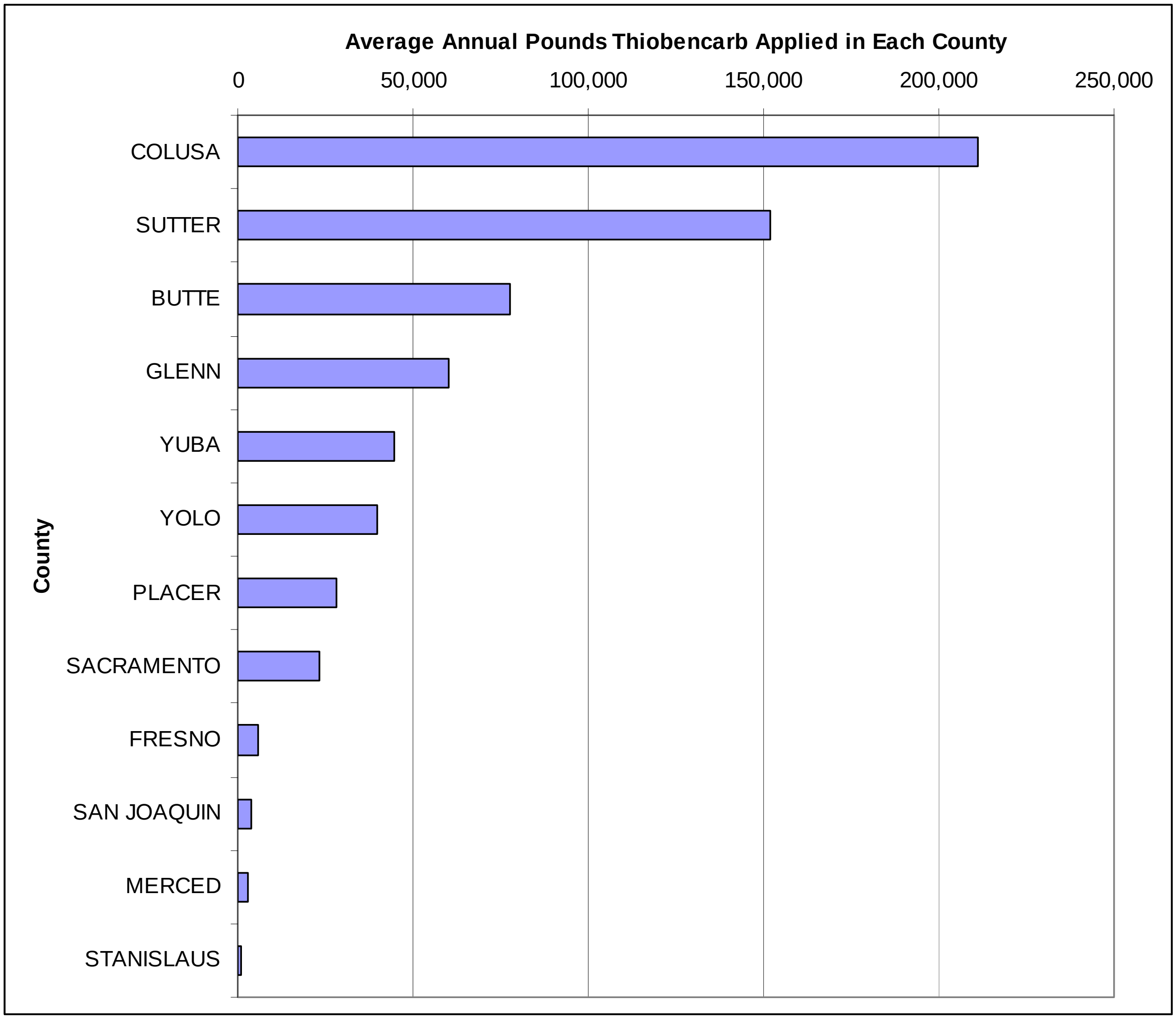

Figure 2-3. Average Annual Pounds Thiobencarb Applied in Each County for the Years 1999-2006. Counties applying a maximum of more than 1000 pounds per year were included in the figure. See Appendix E for additional information.

Table 2-6. Summary of California Department of Pesticide Registration (CDPR) Pesticide Use Reporting (PUR) Data from 1999 to 2006 for Currently Registered Thiobencarb Use on Rice[1]

County	Average Annual Pounds Applied	Average Application Rate (lbs ai/acre)
COLUSA	179,406	3.8
BUTTE	139,591	3.9
SUTTER	130,274	3.8
GLENN	74,925	4.0
YOLO	49,389	3.9
YUBA	18,537	3.4

County	Average Annual Pounds Applied	Average Application Rate (lbs ai/acre)
SACRAMENTO	19,241	3.8
FRESNO	9,618	4.0
PLACER	3,423	3.1
SAN JOAQUIN	5,071	5.5
MERCED	4,283	3.8
STANISLAUS	2,053	4.9*
MADERA	49	4.0
BUTTE	106	4.0
TEHAMA	34	4.0

1 Based on data supplied by BEAD (Carter and Kaul, 2009).

2.5. Assessed Species

Table 2-7 provides a summary of the current distribution, habitat requirements, and life history parameters for the listed species being assessed. Both of the species being assessed have designated critical habitat. More detailed life-history and distribution information can be found in Attachment 1 and Attachment 3. See Figure 2-4 and Figure 2-5 for a map of the current range and designated critical habitat, if applicable, of the assessed listed species.

Table 2-7. Summary of Current Distribution, Habitat Requirements, and Life History Information for the Assessed Listed Species[1]

Assessed Species	Size	Current Range	Habitat Type	Reproductive Cycle	Diet
California red-legged frog (*Rana aurora draytonii*)	Adult (85-138 cm in length), Females – 9-238 g, Males – 13-163 g; Juveniles (40-84 cm in length)	Northern CA coast, northern Transverse Ranges, foothills of Sierra Nevada, and in southern CA south of Santa Barbara	Freshwater perennial or near-perennial aquatic habitat with dense vegetation; artificial impoundments; riparian and upland areas	Breeding: Nov. to Apr. Tadpoles: Dec. to Mar. Young juveniles: Mar. to Sept.	Aquatic-phase[2]: algae, freshwater aquatic invertebrates Terrestrial-phase: aquatic and terrestrial invertebrates, small mammals, fish and frogs
Delta smelt (*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 18 parts per thousand). They live along the freshwater edge of the mixing zone (saltwater-freshwater interface).	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 eat planktonic copepods, cladocerans, amphipods, and insect larvae. Larvae feed on phytoplankton; juveniles feed on zooplankton.

[1] For more detailed information on the distribution, habitat requirements, and life history information of the assessed listed species, see Attachment 1 and Attachment 3.

[2] For the purposes of this assessment, tadpoles and submerged adult frogs are considered "aquatic" because exposure pathways in the water are considerably different than those that occur on land.

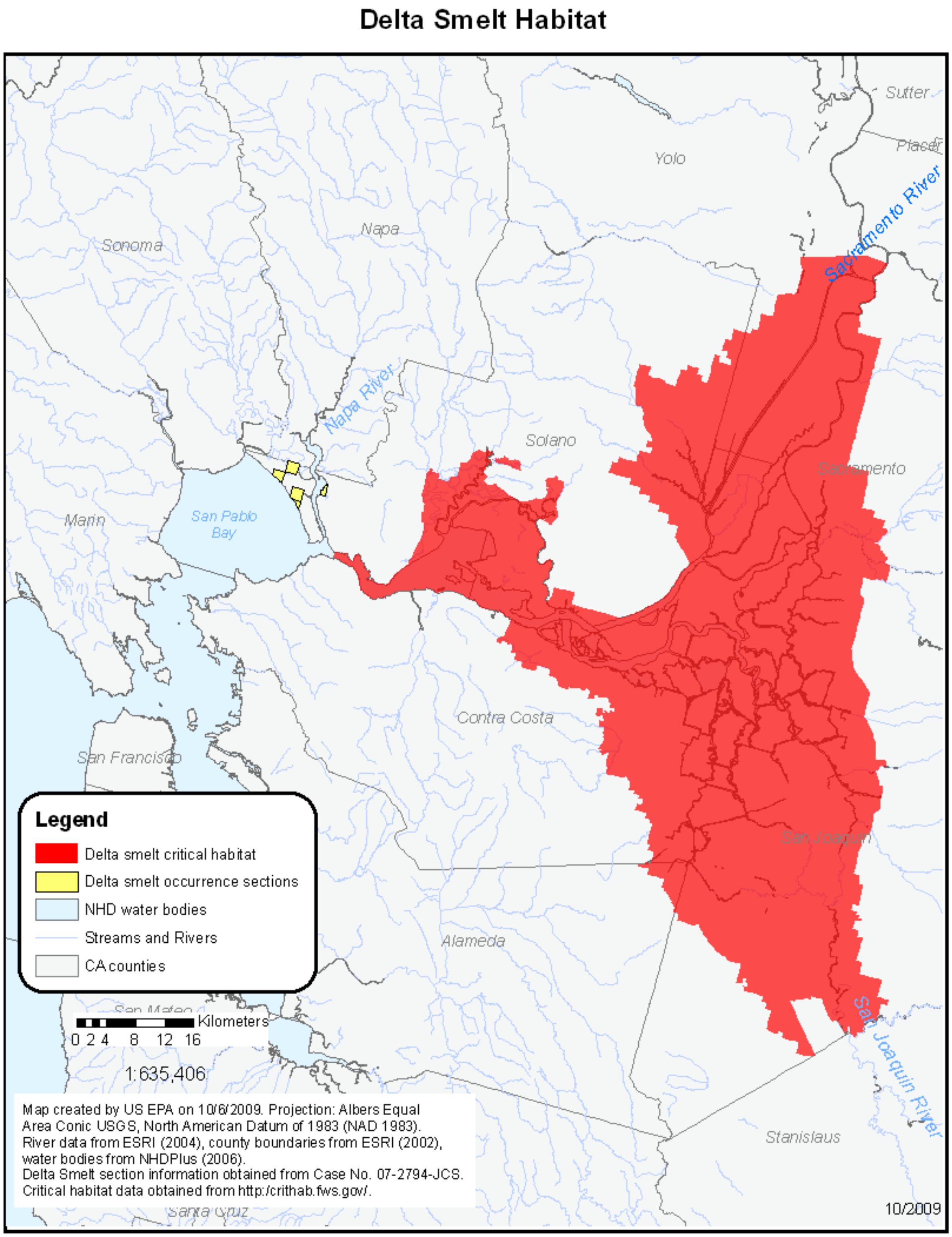

Figure 2-4. Delta smelt critical habitat (USFWS, 2009) and Occurrence Sections identified in Case No. 07-2794-JCS.

Figure 2-5. Recovery Unit, Core Area, Critical Habitat, and Occurrence Designations for CRLF.

2.6. Designated Critical Habitat

Critical habitat has been designated for the CRLF and the DS. '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 receives protection under Section 7 of the ESA through prohibition against destruction or adverse modification with regard to actions carried out, funded, or authorized by a federal Agency. Section 7 requires consultation on federal actions that are likely to result in the destruction or adverse modification of critical habitat.

To be included in a critical habitat designation, the habitat must be '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)). PCEs include, but are not limited to, 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. Table 2-8 describes the PCEs for the critical habitats designated for the CRLF and the DS.

Table 2-8. Designated Critical Habitat PCEs for the CRLF and DS.[1]

Species, Reference	Primary Constituent Elements (PCEs)
CRLF, 50 CFR 414.12(b), 2006	Alteration of channel/pond morphology or geometry and/or increase in sediment deposition within the stream channel or pond.
	Alteration in water chemistry/quality including temperature, turbidity, and oxygen content necessary for normal growth and viability of juvenile and adult CRLFs and their food source.
	Alteration of other chemical characteristics necessary for normal growth and viability of CRLFs and their food source.
	Reduction and/or modification of aquatic-based food sources for pre-metamorphs (*e.g.*, algae)
	Elimination and/or disturbance of upland habitat; ability of habitat to support food source of CRLFs: Upland areas within 200 ft of the edge of the riparian vegetation or dripline surrounding aquatic and riparian habitat that are comprised of grasslands, woodlands, and/or wetland/riparian plant species that provides the CRLF shelter, forage, and predator avoidance
	Elimination and/or disturbance of dispersal habitat: Upland or riparian dispersal habitat within designated units and between occupied locations within 0.7 mi of each other that allow for movement between sites including both natural and altered sites which do not contain barriers to dispersal
	Reduction and/or modification of food sources for terrestrial phase juveniles and adults
	Alteration of chemical characteristics necessary for normal growth and viability of juvenile and adult CRLFs and their food source.
Delta Smelt, 59 FR 65256 65279, 1994	Spawning Habitat—shallow, fresh or slightly brackish backwater sloughs and edge waters 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).
	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 parts per thousand 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.

Species, Reference	Primary Constituent Elements (PCEs)
	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 because these processes are not biologically mediated and, therefore, are not relevant to the endpoints included in this assessment.

More detail on the designated critical habitat applicable to this assessment can be found in Attachment 1 (for the CRLF) and Attachment 3 (for the DS). 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 thiobencarb that may alter the PCEs of the designated critical habitat for the CRLF and DS 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 thiobencarb is expected to directly impact living organisms within the action area, critical habitat analysis for thiobencarb 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

For listed species assessment purposes, the action area is considered to be the area affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02). It is recognized that the overall action area for the national registration of thiobencarb is likely to encompass considerable portions of the United States based on the areas where rice is grown. 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 CRLF and DS and their designated critical habitat within the state of California.

The definition of action area requires a stepwise approach that begins with an understanding of the federal action. The federal action is defined by the currently labeled uses for thiobencarb. An analysis of labeled uses and review of available product labels was completed. Several of the currently labeled uses are special local needs (SLN) uses or are restricted to specific states and are excluded from this assessment. The analysis indicates that for thiobencarb, use on rice is considered as part of the federal action evaluated in this assessment.

Following a determination of the assessed uses, an evaluation of the potential "footprint" of thiobencarb use patterns (*i.e*., the area where pesticide application occurs) 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. A map representing all the land cover types (*e.g*., cultivated crops representing use on rice) that make up the initial area of concern for thiobencarb is presented in Figure 2-6.

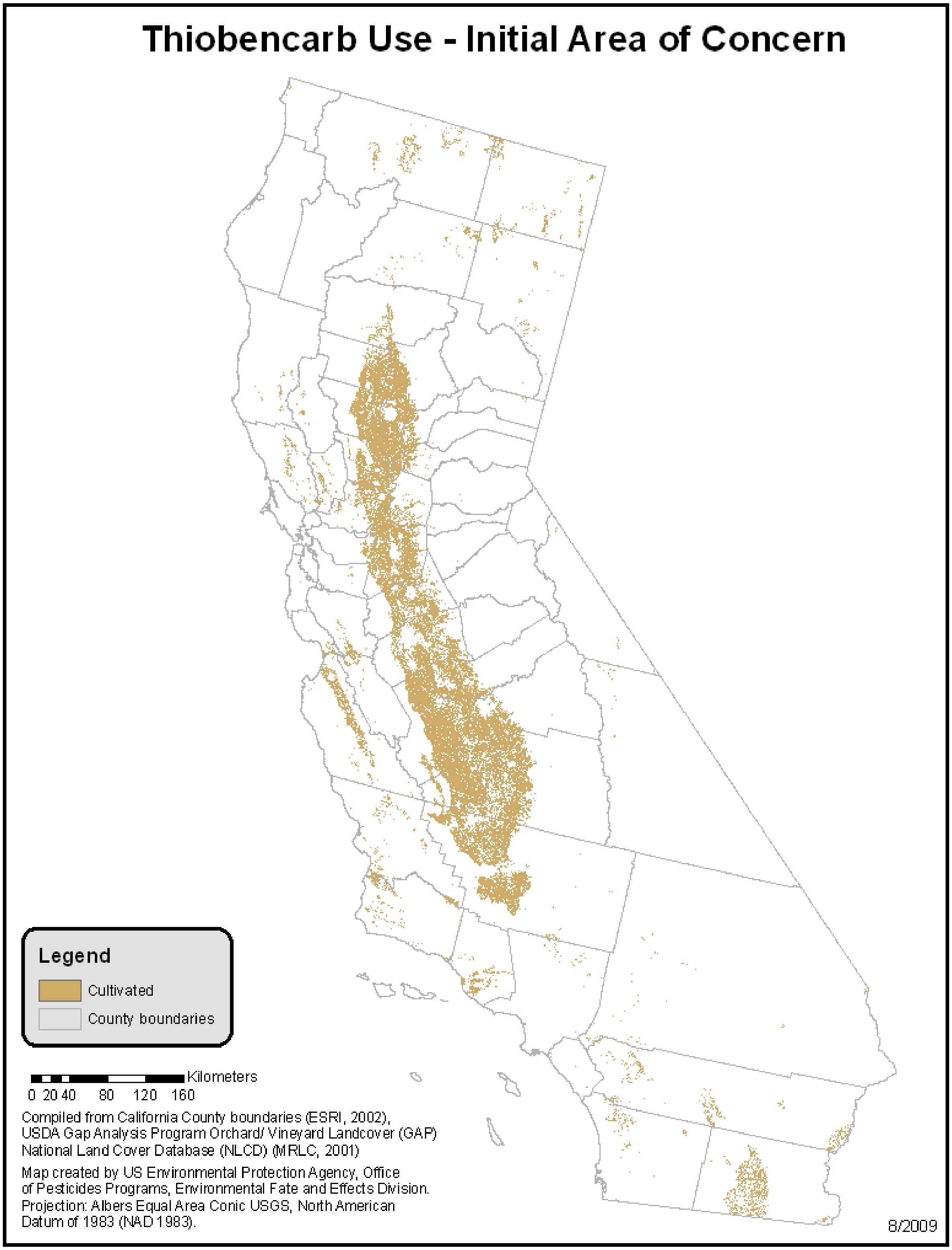

Figure 2-6. Initial area of concern, or "footprint" of potential use, for thiobencarb.

Once the initial area of concern is defined, the next step is to define the potential boundaries of the action area by determining the extent of offsite transport via spray drift and runoff where exposure of one or more taxonomic groups to the pesticide exceeds the listed species LOCs.

The Agency's approach to defining the action area under the provisions of the Overview Document (USEPA, 2004) considers the results of the risk assessment process to establish boundaries for that action area with the understanding that exposures below the Agency's defined Levels of Concern (LOCs) constitute a no-effect threshold. Deriving the geographical extent of this portion of the action area is based on consideration of the types of effects that thiobencarb may be expected to have on the environment, the exposure levels to thiobencarb that are associated with those effects, and the best available information concerning the use of thiobencarb and its fate and transport within the state of California. 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. Therefore, the action area extends to a point where environmental exposures are below any measured lethal or sublethal effect threshold for any biological entity at the whole organism, organ, tissue, and cellular level of organization. In situations where it is not possible to determine the threshold for an observed effect, the action area is not spatially limited and is assumed to be the entire state of California.

Due to the lack of a defined no effect concentration for the most sensitive reported effect and/or a positive result in a mutagenicity test, the spatial extent of the action area (*i.e.*, the boundary where exposures and potential effects are less than the Agency's LOC) for thiobencarb cannot be determined.[6],[7] Therefore, it is assumed that the action area encompasses the entire state of California, regardless of the spatial extent (*i.e.*, initial area of concern or footprint) of the pesticide use(s).

An evaluation of usage information was conducted to determine the area where use of thiobencarb may impact the assessed species. This analysis is used to characterize where predicted exposures are most likely to occur, but does not preclude use in other portions of the action area. A more detailed review of the county-level use information was also completed. These data suggest that thiobencarb is used on rice in California and that rice is grown in areas where the CRLF and DS are commonly found.

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[6] A life-cycle Sheepshead minnow study had significant effects observed at all test concentrations (MRID 00079112).

[7] Thiobencarb did not show mutagenicity in three (Ames assay, dominant lethal assay in mice, and clastogenicity test in human lymphocytes) of four mutagenicity tests conducted (Appendix L). In a micronucleus test in mice, statistically significant increases in the incidence of micronuclei were observed in both sexes (MRID 40352402).

2.8. Assessment Endpoints and Measures of Ecological Effect

Assessment endpoints are defined as "explicit expressions of the actual environmental value that is to be protected."[8] Selection of the assessment endpoints is based on valued entities (*e.g.,* CRLF and DS, organisms important in the life cycle of the assessed species, and the PCEs of its designated critical habitat), the ecosystems potentially at risk (*e.g.,* waterbodies, riparian vegetation, and upland and dispersal habitats), the transport pathways of thiobencarb (*e.g.,* runoff, spray drift, *etc*.), and the routes by which ecological receptors are exposed to thiobencarb *(e.g.,* direct contact, *etc.).*

2.8.1. Assessment Endpoints

Assessment endpoints for the CRLF and DS include direct toxic effects on the survival, reproduction, and growth of individuals, 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 potential effects to PCEs, which are components of the habitat areas that provide essential life cycle needs of the assessed species. Each assessment endpoint requires one or more "measures of ecological effect," defined as changes in the attributes of an assessment endpoint or changes in a surrogate entity or attribute in response to exposure to a pesticide. Specific measures of ecological effect are generally evaluated based on acute and chronic toxicity information from registrant-submitted guideline tests that are performed on a limited number of organisms. Additional ecological effects data from the open literature are also considered. It should be noted that assessment endpoints are limited to direct and indirect effects associated with survival, growth, and fecundity, and do not include the full suite of sublethal effects used to define the action area. According the Overview Document (USEPA, 2004), the Agency relies on acute and chronic effects endpoints that are either direct measures of impairment of survival, growth, or fecundity or endpoints for which there is a scientifically robust, peer reviewed relationship that can quantify the impact of the measured effect endpoint on the assessment endpoints of survival, growth, and fecundity.

A discussion of 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. A summary of the assessment endpoints and measures of ecological effect selected to characterize potential assessed direct and indirect risks for each of the assessed species associated with exposure to thiobencarb is provided in Section 2.5 and Table 2-9.

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 aquatic-phase amphibians, freshwater fish, freshwater invertebrates, E/M fish, invertebrates, aquatic plants, birds (surrogate for terrestrial-phase amphibians), 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 thiobencarb.

[8] From U.S. EPA (1992). *Framework for Ecological Risk Assessment.* EPA/630/R-92/001.

Table 2-9 identifies the taxa used to assess the potential for direct and indirect effects from the uses of thiobencarb 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 2-10.

Table 2-9. Taxa Used in the Analyses of Direct and Indirect Effects for the Assessed Listed Species.

Listed Species	Birds/ Terr. Amphibian	Mammals	Terr. Plants	Terr. Inverts.	FW Fish and Aquatic Phase Amphibians	FW Inverts.	E/M Fish	Estuarine /Marine Inverts.	Aquatic Plants
California red-legged frog	Direct Indirect (prey)	Indirect (prey)	Indirect (habitat)	Indirect (prey)	Direct Indirect (prey)	Indirect (prey)	N/A	N/A	Indirect (food/ habitat)
Delta smelt	N/A	N/A	Indirect (habitat)	N/A	Direct	Indirect (prey)	Direct	Indirect (prey)	Indirect (food/ habitat)

Abbreviations: N/A = Not applicable; Terr. = Terrestrial; Invert. = Invertebrate; FW = Freshwater; E/M=Estuarine/marine

1 The most sensitive species across freshwater and saltwater environments was used to assess effects for the DS.

Table 2-10. Taxa and Assessment Endpoints Used to Evaluate the Potential for Use of Thiobencarb to Result in Direct and Indirect Effects to the Assessed Listed Species.

Taxa Used to Assess Direct and/or Indirect Effects to Assessed Species	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
1. Freshwater Fish and Aquatic-phase Amphibians	Direct Effect – -Aquatic-phase CRLF -DS	Survival, growth, and reproduction of individuals via direct effects	1a. Amphibian acute LC_{50} (ECOTOX) or most sensitive fish acute LC_{50} (guideline or ECOTOX) if no suitable amphibian data are available 1b. Amphibian chronic NOAEC (ECOTOX) or most sensitive fish chronic NOAEC (guideline or ECOTOX) 1c. Amphibian early-life stage data (ECOTOX) or most sensitive fish early-life stage NOAEC (guideline or ECOTOX)
	Indirect Effect (prey) -Aquatic and terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, fish and aquatic-phase amphibians)	
2. Freshwater Invertebrates	Indirect Effect (prey) -Aquatic and terrestrial-phase CRLF -DS	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, freshwater invertebrates)	2a. Most sensitive freshwater invertebrate EC_{50} (guideline or ECOTOX) 2b. Most sensitive freshwater invertebrate chronic NOAEC (guideline or ECOTOX)
3. Estuarine/Marine Fish	Direct Effect - - DS Indirect Effect (prey) -DS	Survival, growth, and reproduction of individuals via indirect effects on aquatic prey food supply (*i.e.*, E/M fish)	3a. Most sensitive E/M fish EC_{50} (guideline or ECOTOX) 3b. Most sensitive E/M fish chronic NOAEC (guideline or ECOTOX)
4. Estuarine/Marine Invertebrates	Indirect Effect (prey)	Survival, growth, and reproduction of individuals	4a. Most sensitive E/M invertebrate EC_{50} (guideline or ECOTOX)

Taxa Used to Assess Direct and/or Indirect Effects to Assessed Species	Assessed Listed Species	Assessment Endpoints	Measures of Ecological Effects
		via indirect effects on aquatic prey food supply (*i.e.*, estuarine/marine invertebrates)	4b. Most sensitive E/M invertebrate chronic NOAEC (guideline or ECOTOX)
5. Aquatic Plants (freshwater/marine)	<u>Indirect Effect (food/habitat)</u> -Aquatic-phase CRLF - DS	Survival, growth, and reproduction of individuals 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	<u>Direct Effect</u> -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via direct effects	6a. Most sensitive bird[b] or terrestrial-phase amphibian acute LC_{50} or LD_{50} (guideline or ECOTOX) 6b. Most sensitive bird[b] or terrestrial-phase amphibian chronic NOAEC (guideline or ECOTOX)
	<u>Indirect Effect (prey)</u> -CRLF	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (birds)	
7. Mammals	<u>Direct Effect</u> -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via direct effects	7a. Most sensitive laboratory rat acute LC_{50} or LD_{50} (guideline or ECOTOX) 7b. Most sensitive laboratory rat chronic NOAEC (guideline or ECOTOX)
	<u>Indirect Effect (prey)</u> -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via indirect effects on terrestrial prey (mammals)	
8. Terrestrial Invertebrates	<u>Indirect Effect (prey)</u> -Terrestrial-phase CRLF	Survival, growth, and reproduction of individuals via direct effects	8a. Most sensitive terrestrial invertebrate acute EC_{50} or LC_{50} (guideline or ECOTOX)[c] 8b. Most sensitive terrestrial invertebrate chronic NOAEC (guideline or ECOTOX)
		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> -Aquatic and terrestrial-phase CRLF - DS	Survival, growth, and reproduction of individuals 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} for dicots (seedling emergence, vegetative vigor, or ECOTOX)
	<u>Indirect Effect (food/habitat) (obligate relationship)</u> -Terrestrial and aquatic phase CRLF - DS		

2.8.2. Assessment Endpoints for Designated Critical Habitat

As previously discussed, designated critical habitat is assessed to evaluate actions related to the use of thiobencarb 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 thiobencarb 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 thiobencarb to the environment. The following risk hypotheses are presumed in this assessment:

The labeled use of thiobencarb within the action area may:

- directly affect the CRLF and/or DS by causing mortality or by adversely affecting growth or fecundity;
- indirectly affect the CRLF and/or DS and/or modify their designated critical habitat by reducing or changing the composition of food supply;
- indirectly affect the CRLF and/or DS 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 the CRLF and/or DS 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 the CRLF and/or DS 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).

2.9.2. Diagram

The conceptual model is a graphic representation of the structure of the risk assessment. It specifies the thiobencarb release mechanisms, biological receptor types, and effects endpoints of potential concern. The conceptual models for aquatic and terrestrial phases of the CRLF and DS and the conceptual models for the aquatic and terrestrial PCE components of critical habitat are shown in Figure 2-7 and Figure 2-8. 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 will be 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 the CRLF and DS and modification to designated critical habitat is expected to be negligible.

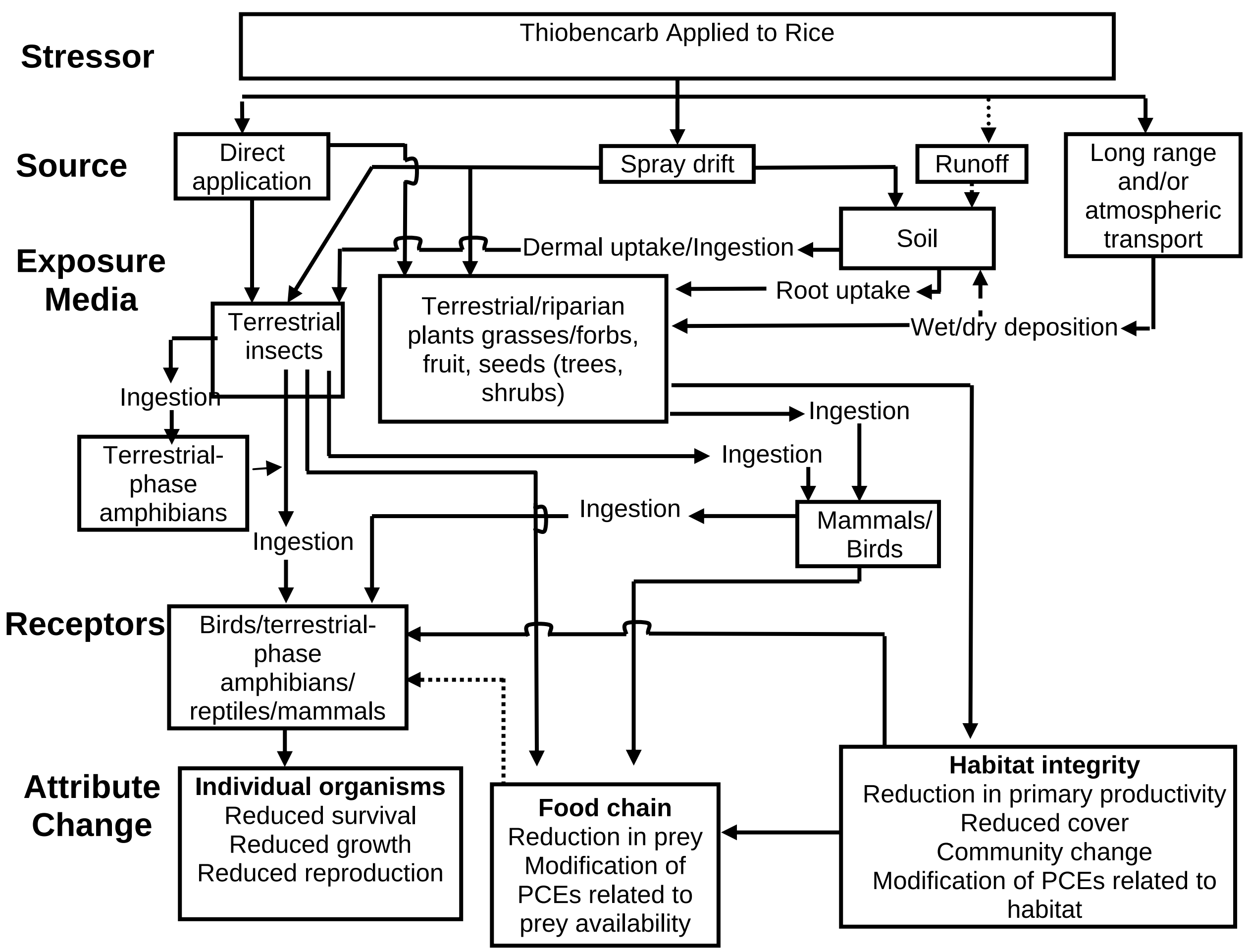

Figure 2-7. Conceptual Model for Terrestrial-Phase of the Assessed Species.

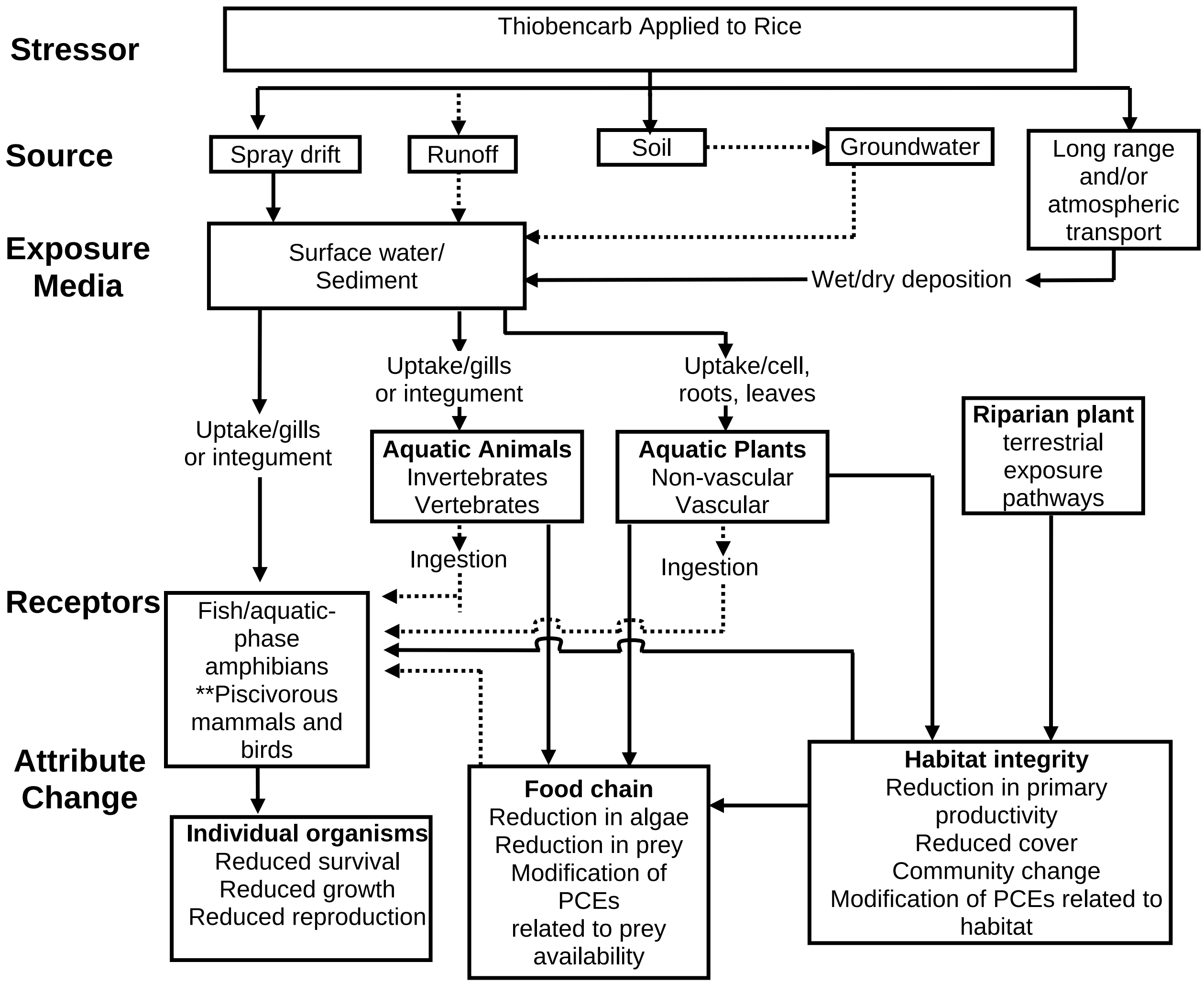

Figure 2-8. Conceptual Model for Aquatic-Phase of the Assessed Species.

2.10. Analysis Plan

In order to address the risk hypothesis, the potential for direct and indirect effects to the CRLF and DS, prey items, and habitat is estimated based on a taxon-level approach. In the following sections, the use, environmental fate, and ecological effects of thiobencarb 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 thiobencarb is estimated using the probit dose-response slope and either the level of concern (discussed below) or actual calculated risk quotient value.

2.10.1. Measures of Exposure

The environmental fate properties of thiobencarb along with available monitoring data indicate that water and sediment discharge from rice paddies and spray drift are the principle potential transport mechanisms of thiobencarb to the aquatic and terrestrial habitats. Thiobencarb is also semi-volatile (*e.g*., falls between nonvolatile and intermediate to high volatility) and may travel to nearby fields in the air or to remote areas via long range transport (based on volatility classification for vapor pressure in OPPTS Guideline 835.6100 and monitoring data). In this assessment, transport of thiobencarb in rice paddy water, and spray drift is considered in deriving quantitative estimates of thiobencarb exposure to CRLF and DS and their prey and habitats. Additionally, exposure due to deposition of thiobencarb in precipitation and movement of thiobencarb into ground water were qualitatively assessed. Some bioaccumulation may occur; however, depuration of thiobencarb is almost complete after 30 days and the log K_{OW} of thiobencarb is below four (see Section 2.4.1.3g) (Ceesay, 2002). Therefore, we do not expect bioaccumulation of thiobencarb to be a major exposure pathway for the CRLF and DS.

Measures of exposure are based on aquatic and terrestrial models that predict estimated environmental concentrations (EECs) of thiobencarb using maximum labeled application rates and methods of application. The models used to predict aquatic EECs are the Tier I Rice Model and Tier I Rice Model modified to include aquatic dissipation. Monitoring results were used to estimate chronic exposure to the DS. The model used to predict terrestrial EECs on food items is T-REX. The model used to derive EECs relevant to terrestrial and wetland plants is TerrPlant. These models are parameterized using relevant reviewed registrant-submitted environmental fate data.

2.10.1.a. Estimating Exposure in the Aquatic Environment

Concentrations of thiobencarb in surface water were estimated using the standard Tier I Rice Model and a modified version of the Tier I Rice Model that accounts for possible dissipation in the paddy water.

The original Tier I Rice Model relies on an equilibrium partitioning concept to provide conservative screening estimates of EECs resulting from application of pesticides to rice paddies. When a pesticide is applied to a rice paddy, the model assumes that it will instantaneously partition between a water phase and a sediment phase. The Tier I Rice Model was calibrated to generate estimates that are similar to or greater than dissolved concentrations measured within rice paddies and in discharged paddy water. The model does not account for pesticide degradation, volatilization, dilution, or other dissipation processes. The sediment interaction depth was determined by calibrating the model to maximum residues measured in paddy water in dissipation studies. Pesticide degradation, mass transfer, volatilization, dilution and other dissipation processes may have occurred in those datasets but probably had little affect on the calibration because the model was calibrated to the maximum measured residues. The model was not evaluated or calibrated for concentrations measured in sediment and does not account for residues bound to suspended sediment. Guidance for using the Tier I Rice Model may be found on the U.S. Environmental Protection Agency (EPA) Water Models web-page, see http://www.epa.gov/oppefed1/models/water/#rice (Bradbury, 2007).

The Tier I Rice Model was provisionally modified to account for dissipation in the paddy water during the 14 day holding period required prior to water release from the paddy. Rates of dissipation were based on the slowest dissipation rate observed in aquatic dissipation studies. Assumptions of the Tier I Rice Model, other than stability to dissipation and degradation, apply to the modified model.

2.10.1.b. Estimating Exposure in the Terrestrial Environment

Exposure estimates for the terrestrial animals assumed to be in the target area or in an area exposed to spray drift are derived using the T-REX model (version 1.4.1, 10/08/2009). This model incorporates the Kenaga nomograph, as modified by Fletcher *et al.* (1994), which is based on a large set of actual field residue data (Fletcher *et al.*, 1994). The upper limit values from the nomograph represented the 95th percentile of residue values from actual field measurements (Hoerger and Kenaga, 1972).

For modeling purposes, direct exposures of the CRLF and DS to thiobencarb through contaminated food are estimated using the EECs for the small bird (20 g) which consumes small insects. Dietary-based and dose-based exposures of potential prey (small mammals) are assessed using the small mammal (15 g) which consumes short grass. The small bird (20g) consuming small insects and the small mammal (15g) consuming short grass are used because these categories represent the largest RQs of the size and dietary categories in T-REX that are appropriate surrogates for the CRLF and one of its prey items. Estimated exposures of terrestrial insects to thiobencarb are bound by using the dietary based EECs for small insects and large insects.

Birds are currently used as surrogates for terrestrial-phase amphibians and reptiles. However, amphibians and reptiles are poikilotherms (body temperature varies with environmental temperature) while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). Therefore, amphibians and reptiles 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 and reptiles on a daily dietary intake basis, assuming similar caloric content of the food items. Therefore, the use of avian food intake allometric equation as a surrogate to amphibians and reptiles is likely to result in an over-estimation of exposure and risk for reptiles and terrestrial-phase amphibians. Therefore, T-REX (version 1.3.1) has been refined to the T-HERPS model (v. 1.0), which allows for an estimation of food intake for poikilotherms using the same basic procedure as T-REX to estimate avian food intake.

EECs for terrestrial plants inhabiting dry and wetland areas are derived using TerrPlant (version 1.2.2, 12/26/2006). This model uses estimates of pesticides in runoff and in spray drift to calculate EECs. EECs are based upon solubility, application rate and minimum incorporation depth. Runoff is not expected to commonly occur in rice paddies; therefore, only the exposure estimated based on spray drift are used in the assessment.

Spray drift models, AGDISP and/or AgDRIFT are used to assess exposures of terrestrial animals to thiobencarb deposited on terrestrial habitats by spray drift. In addition to the buffered area from the spray drift analysis, the downstream extent of thiobencarb that exceeds the LOC for the effects determination is also considered.

2.10.2. Measures of Effect

Data identified in Section 2.8 are used as measures of effect for direct and indirect effects to the CRLF and DS. Data were obtained from registrant submitted studies or from literature studies identified by ECOTOX. The ECOTOXicology database (ECOTOX) was searched in order to provide more ecological effects data and in an attempt to bridge existing data gaps. ECOTOX is a source for locating single chemical toxicity data for aquatic life, terrestrial plants, and wildlife. ECOTOX was created and is maintained by the USEPA, Office of Research and Development, and the National Health and Environmental Effects Research Laboratory's Mid-Continent Ecology Division. Each publication abstracted for the ECOTOX database effort is assigned a unique reference number. The number is an E followed by a five digit number and is used to identify ECOTOX references in this document.

The assessment of risk for direct effects to the terrestrial-phase CRLF makes the assumption that toxicity of thiobencarb to birds is similar to or less than the toxicity to terrestrial-phase amphibians and reptiles (this also applies to potential prey items). The same assumption is made for fish and aquatic-phase CRLF (again, this also applies to potential prey items). Data on aquatic phase amphibians will be used qualitatively in the risk assessment.

The acute measures of effect used for animals in this screening level assessment are the LD_{50}, LC_{50} and EC_{50}. LD stands for "Lethal Dose", and LD_{50} is the amount of a material, given all at once, that is estimated to cause the death of 50% of the test organisms. LC stands for “Lethal Concentration” and LC_{50} is the concentration of a chemical that is estimated to kill 50% of the test organisms. EC stands for “Effective Concentration” and the EC_{50} is the concentration of a chemical that is estimated to produce a specific effect in 50% of the test organisms. Endpoints for chronic measures of exposure for listed and non-listed animals are the NOAEL/NOAEC and NOAEC. NOAEL stands for “No Observed-Adverse-Effect-Level” and refers to the highest tested dose of a substance that has been reported to have no harmful (adverse) effects on test organisms. The NOAEC (*i.e.*, “No-Observed-Adverse-Effect-Concentration”) is the highest test concentration at which none of the observed effects were statistically different from the control. The NOEC is the No-Observed-Effects-Concentration. For non-listed plants, only acute exposures are assessed (*i.e.*, EC_{25} for terrestrial plants and EC_{50} for aquatic plants).

It is important to note that the measures of effect for direct and indirect effects to the assessed species and their designated critical habitat are associated with impacts to survival, growth, and fecundity, and do not include the full suite of sublethal effects used to define the action area. According the Overview Document (USEPA 2004), the Agency relies on effects endpoints that are either direct measures of impairment of survival, growth, or fecundity or endpoints for which there is a scientifically robust, peer reviewed relationship that can quantify the impact of the measured effect endpoint on the assessment endpoints of survival, growth, and fecundity.

2.10.2.a. 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 thiobencarb, and the likelihood of direct and indirect effects to CRLF and DS 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. For the assessment of thiobencarb risks, 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 levels of concern (LOCs) (USEPA, 2004)(see Appendix F).

For this endangered species assessment, listed species LOCs are used for comparing RQ values for acute and chronic exposures of thiobencarb directly to the CRLF and DS. If estimated exposures directly to the assessed species of thiobencarb resulting from a particular use are sufficient to exceed the listed species LOC, then the effects determination for that use is "may affect". When considering indirect effects to the assessed species due to effects to prey, the listed species LOCs are also used. If estimated exposures to the prey of the assessed species of thiobencarb resulting from a particular use are sufficient to exceed the listed species LOC, then the effects determination for that use is a "may affect." If the RQ being considered also exceeds the non-listed species acute risk LOC, then the effects determination is a LAA. If the acute RQ is between the listed species LOC and the non-listed acute risk species LOC, then further lines of evidence (*i.e*. probability of individual effects, species sensitivity distributions) are considered in distinguishing between a determination of NLAA and a LAA. If the RQ being considered for a particular use exceeds the non-listed species LOC for plants, the effects determination is "may affect". Further information on LOCs is provided in Appendix F.

2.10.2.b. 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 thiobencarb 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.

3. Exposure Assessment

Thiobencarb is formulated as an emulsifiable concentrate and as granules. It may be applied via ground or aerial application methods. Risks from ground boom and aerial applications are considered in this assessment because they are expected to result in the highest off-target levels of thiobencarb due to generally higher spray drift levels. Ground boom and aerial modes of application tend to use lower volumes of application applied in finer sprays than applications coincident with sprayers and spreaders and thus have a higher potential for off-target movement via spray drift.

3.1. Aquatic Exposure Assessment

The estimated surface water environmental concentrations (EEC) presented here are based on the Tier 1 Rice Model, which assumes flooded fields (wet seeding) and on Aquatic Field Dissipation studies. Based on the environmental fate data for thiobencarb, input parameters used for the Tier 1 Rice Model are shown in Table 3-1.

Table 3-1. Chemical Specific Input Parameters for Thiobencarb

Parameter	Input Value and Unit	Source
Maximum application rate	4 lb/acre	Product Labels
Maximum number of applications	1	Product Labels
Partition coefficient K_{oc}[b]	900 mL/g OC (average of 5 values, range 384-1435)	MRID 41215313

The Tier 1 Rice Model v1.0 (May 8, 2007) estimates the peak concentration of pesticide in a 0.1 meter-deep rice paddy, and does not account for any dissipation processes, with the exception of partitioning to sediment. The relevant equation is:

$$Cw = m_{ai}' / (0.00105 + 0.0000013*Koc)$$

Where Cw is the paddy water concentration (ppb), m_{ai}' is the application rate (kg/hectare) and Koc is the organic-carbon normalized partition coefficient. The result is given in Table 3-2.

Table 3-2. Results of Tier 1 Rice Model

Application Rate	Peak EEC, ppb
4 lb/acre (4.48 kg/hectare)	2018.

This Tier I Rice Model exposure estimate was refined by allowing the concentration in water (2,018 ppb) to decay, using a first-order exponential decay equation, at the slower of the two rates from the two available wet-seeded aquatic field dissipation studies (decay rate, 0.1252/day, half-life 5.5 days). The decay rate constants were k=0.1252 (halflife 5.54 days) for MRID 43404005, and k=0.1596 (halflife 4.34 days) for Ross & Sava (1986). See Figure 3-1 for a graph of the result. This analysis shows that at 14 days, the concentration is 350 ppb; at 19 days, 187 ppb, and at 30 days, 47 ppb. A chronic estimate of exposure in the rice paddy was estimated as an average 14-day concentration of 968 µg/L. A 21-day average and 60-day average were not estimated from the Tier 1 Rice model because rice paddy water is expected to be released after 14 days. This approach neglects any thiobencarb that may be adsorbed to sediment, however the exposures calculated in the water column are sufficient to exceed Agency Levels of Concern. Thus, explicit consideration of adsorbed thiobencarb's direct or indirect effects would only strengthen this conclusion.

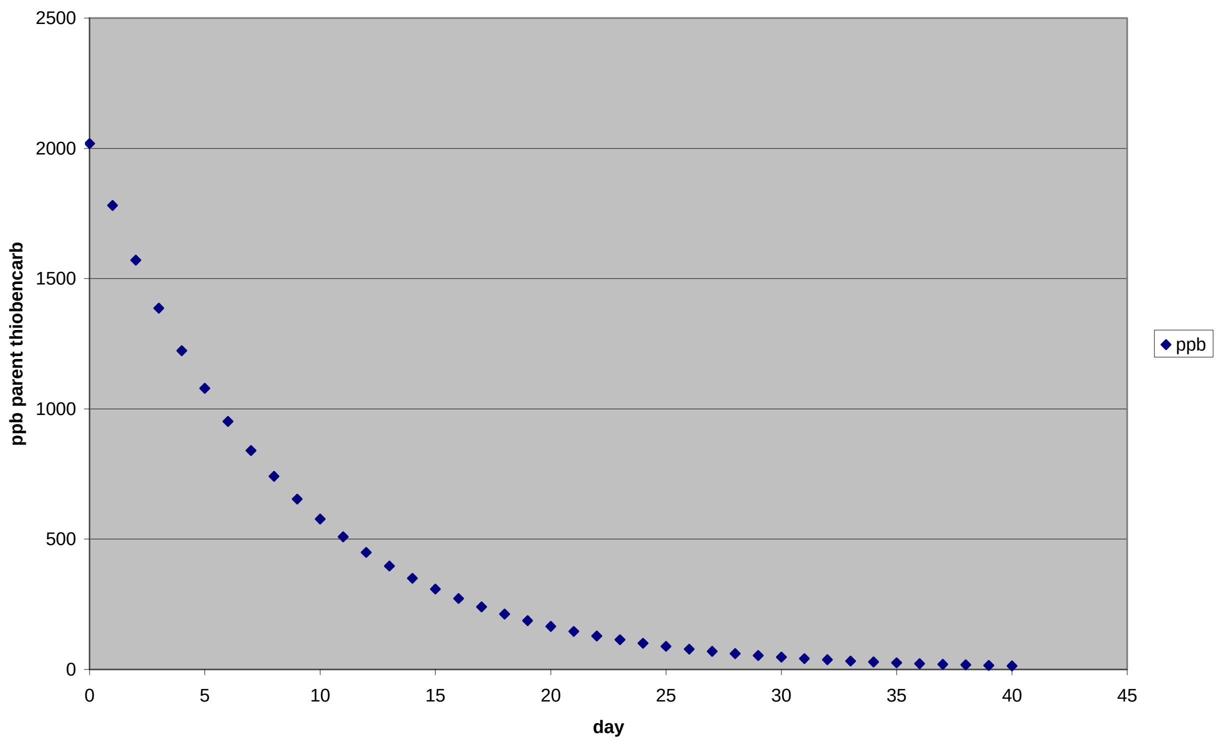

Figure 3-1. Estimated Concentrations of Thiobencarb in Water Starting with the Tier I Rice Model Estimated Initial Concentration and Using the Field Decay Rate from an Aquatic Dissipation Study (decay rate = 0.1252/day, half-life = 5.5 days).

Since the Rice Model cannot provide time-averaged EECs for calculation of chronic RQ for aquatic invertebrate and fish, respectively, a spreadsheet was used to estimate them. For risk estimation, a 14-day average in the rice paddy was calculated using the Tier 1 Rice Model peak concentration (2018 ppb) as a starting point. Concentrations on days 1 to 14 were calculated using a first-order decay model with the slower field dissipation rate (5.5 days), and the results averaged to get the 14-day average EEC for chronic RQ calculation (968 ppb). A 14^{th}-day value (350 ppb) was also calculated for acute exposure characterization on the day of paddy water release

For characterization purposes, 21-day and 60-day chronic concentrations were calculated from the field dissipation studies (MRID 43404005 and Ross & Sava). For each study, concentrations were projected out to day 60 from the peak concentration day (not day one), using the study-specific decay rate. The 21-day or 60-day average was then calculated from day zero to day 21 or day 60. The estimated aquatic EECs are shown in Table 3-3. The values are similar for the two studies: about 205 ppb for the 21-day average and about 75 ppb for the 60-day average. These values are lower than the 14-day average (968 ppb) used in risk estimation and may be used for characterization of chronic risks.

The in-paddy exposure assessment is considered conservative, since it uses the Tier 1 Rice Model, and produces no EECs that are below monitored concentrations. The peak concentration (2018 ppb) is higher than the observed peak in either field dissipation study (438 to 576 ppb). The highest observed concentration in California surface water (170 ppb in 1982 at Colusa Basin Drain #5, which is known to the location most contaminated by rice herbicides in the state) is lower than the concentration calculated by this method for the day of paddy water release (350 ppb on day 14). In addition, the monitoring data are considered to be robust and suitable for exposure assessment in receiving waters.

Table 3-3. Aquatic Exposure using 4 lb/acre (4.48 kg/hectare)

Exposure reference	Peak EEC, ppb	14-Day Average in Paddy Water	14-day Release Value	21 day average	60 day average
Modeled Values In Paddy and When Paddy Water is Released (Risk Estimation)					
Rice Model Tier I (Peak Concentration in Paddy)	2018	---	---	---	---
Tier I Rice Model and California Wet Seeded Rice Aquatic Dissipation Rate (MRID 43404005)	2018	968	350	---	---
Concentrations Observed In Aquatic Dissipation Studies (In Rice Paddy) (Risk Characterization)					
California Wet Seeded Rice Aquatic Dissipation (MRID 43404005)	438	---	---	202	80
Aquatic Dissipation Study (Ross and Sava, 1986)	576	---	---	209	70

In the submitted study (MRID 43404005), the water concentration of parent thiobencarb fell to 107.7 ppb (n=4) on day 10 after application, and 44.2 ppb (n=4) on day 14 after a single application of granular formulation. Including the two measured degradates (thiobencarb sulfoxide and 4-chlorobenzylmethylsulfone), the total water concentrations were 125.8 ppb on day 10, 58.5 ppb on day 14, and 19.3 ppb on day 21. Results from the California wet seeded rice aquatic dissipation study are graphed in Figure 3-2. The granular formulation (Bolero 10G) may have resulted in the delayed peak concentration observed at 3 days after the application.

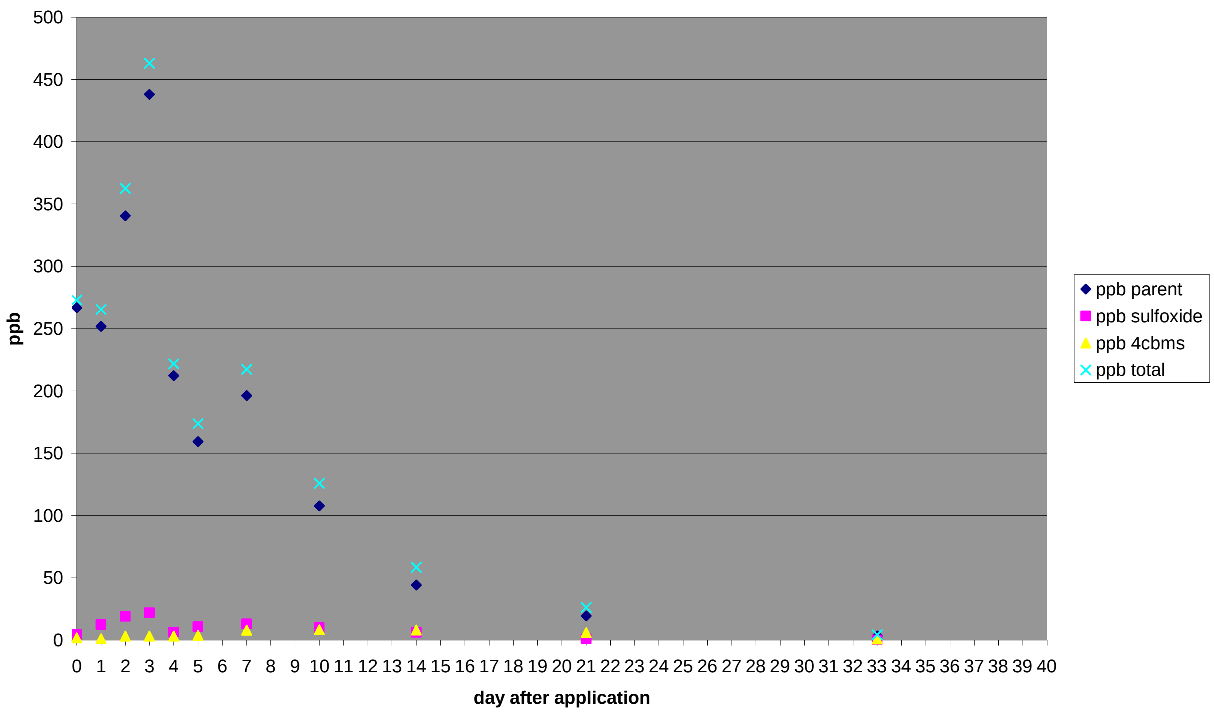

Figure 3-2. Water Concentrations of Thiobencarb and Two Degradates in California Aquatic Dissipation Study on Wet Seeded Rice After Application of Granular Formulation of Thiobencarb (MRID 43404005).[9]

In Ross & Sava (1986), the water concentration of parent thiobencarb after a single application was 367 ppb on day 8, 56 ppb on day 16, and 8 ppb on day 32. No degradates were measured. These data are graphed in Figure 3-3.

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[9] The sulfoxide in the Figure legend stands for thiobencarb sulfoxide and 4cbms stands for 4-chlorobenzylmethylsulfone.

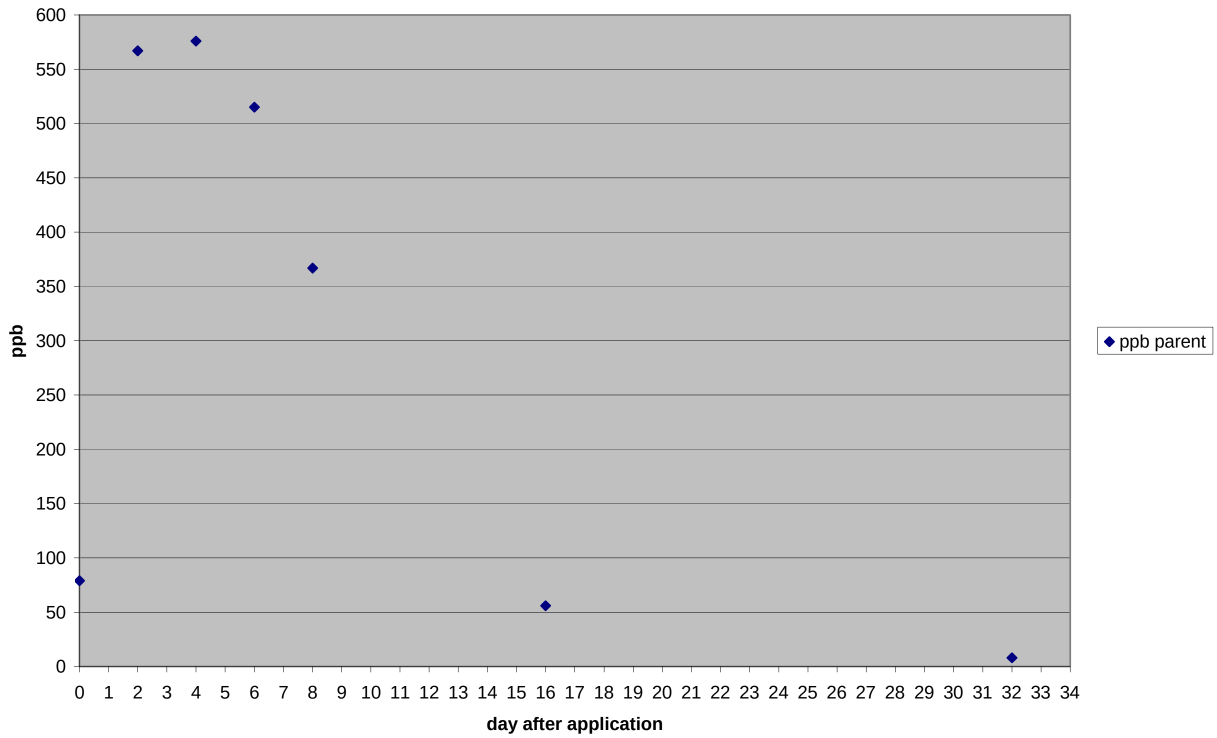

Figure 3-3. Thiobencarb Concentrations in Water in Aquatic Field Study (Ross and Sava, 1986).

3.1.1. 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 thiobencarb data from the USGS National Water Quality Assessment (NAWQA) program (http://water.usgs.gov/nawqa) and data from the CADPR. In addition, atmospheric monitoring data for thiobencarb from the open literature are summarized below.

3.1.1.a. Characterization of Surface Water Monitoring Data

Rice is grown in the Sacramento and San Joaquin river valleys to the north and south of San Francisco Bay. Discharged paddy water containing thiobencarb residues flows through the Delta on its way to the bay. In order to quantify the exposure in time and space, all surface water monitoring results for thiobencarb from the California surface water database and from the NAWQA program were retrieved. Data were also obtained from Orlando and Kuivila (2004). Both the highest exposures and those furthest downstream from the rice growing areas on each river are characterized below. These exposures were calculated to estimate CRLF and DS exposures in habitats other than directly in rice paddies. While CRLF may use rice paddies as habitat, DS will not be exposed to water in the rice paddy as they do not have a way to travel into the rice paddy. They are found primarily below Isleton on the Sacramento River, below Mossdale on the San Joaquin River, and in the Suisan Bay (USDOI, 2008). During spawning (February through June), they may be found in:

> “(1) the Sacramento River as high as Sacramento, (2) the Mokelumne River system, (3) the Cache Slough region, (4) the Delta, (5) Montezuma Slough, (6) Suisun Bay, (7) Suisun Marsh, (8) Carquinez Strait, (9) Napa River, and (10) San Pablo Bay." (USDOI, 2008)

Since 1982, they are primarily found in the northwestern Delta in the channel of the Sacramento River and the Suisan Bay (USDOI, 2008).

The highest concentrations observed in receiving waters occur at Colusa Basin Drain #5 on the Sacramento River. Monitoring results for this site are shown in Figure 3-4. This waterway does not meet state standards for water quality due to pesticide residues, principally from rice production. The highest peak concentration observed since the 1984 establishment of a 14-day holding requirement for paddy water treated with thiobencarb is 37.4 µg/L in 1994. Prior to the holding time requirement, thiobencarb was as high as 170 µg/L in 1982 (Orlando and Kuivila, 2004). This concentration is within a factor of two of the day-14 release water concentration (350 ppb), as calculated by the extended Tier 1 Rice Model (Table 3-3 above).

The highest time-averaged exposure observed at Colusa Basin Drain is 5.37 µg/L, over the period of May 9 to June 22, 2000. This was calculated as the average of 14 samples taken over the 44-day period.

Further downstream, thiobencarb residues are lower, but still detectable. On the Sacramento River at Freeport, the highest recent peak concentration was 0.65 µg/L on May 22, 2002. The chronic exposure at this station was 0.146 µg/L over the period of May 22 to Nov 13, 2002 (based on 5 samples).

On the San Joaquin River, thiobencarb was detected as far downstream as the San Joaquin River at Vernalis station, with a peak concentration of 0.528 µg/L (May 22, 1993), and a chronic concentration of 0.189 µg/L (May 22 to May 30, 1993). At the San Joaquin River at Maze Rd bridge station, a peak concentration of 0.697 µg/L was observed on June 12, 2001.

Each of these NAWQA stations is 20 miles or more upstream from the Suisun Bay and its tributaries. Thus, there may be further dilution of the thiobencarb residues and lower exposure for the DS when it is not spawning. A recent USGS paper (Kuivila and Jennings, 2007) measured thiobencarb at Mallard Island, at the eastern end of Suisun Bay in a tidally-influenced area, 8 km downstream of the confluence of the Sacramento and San Joaquin rivers. Samples were collected daily or twice daily from January to June, 1996. Thiobencarb was found in 28% of the samples with a maximum concentration of 66 ng/L (0.066 µg/L). This is a factor of ten lower than the concentrations measured at Freeport, the Maze Road bridge or Vernalis.

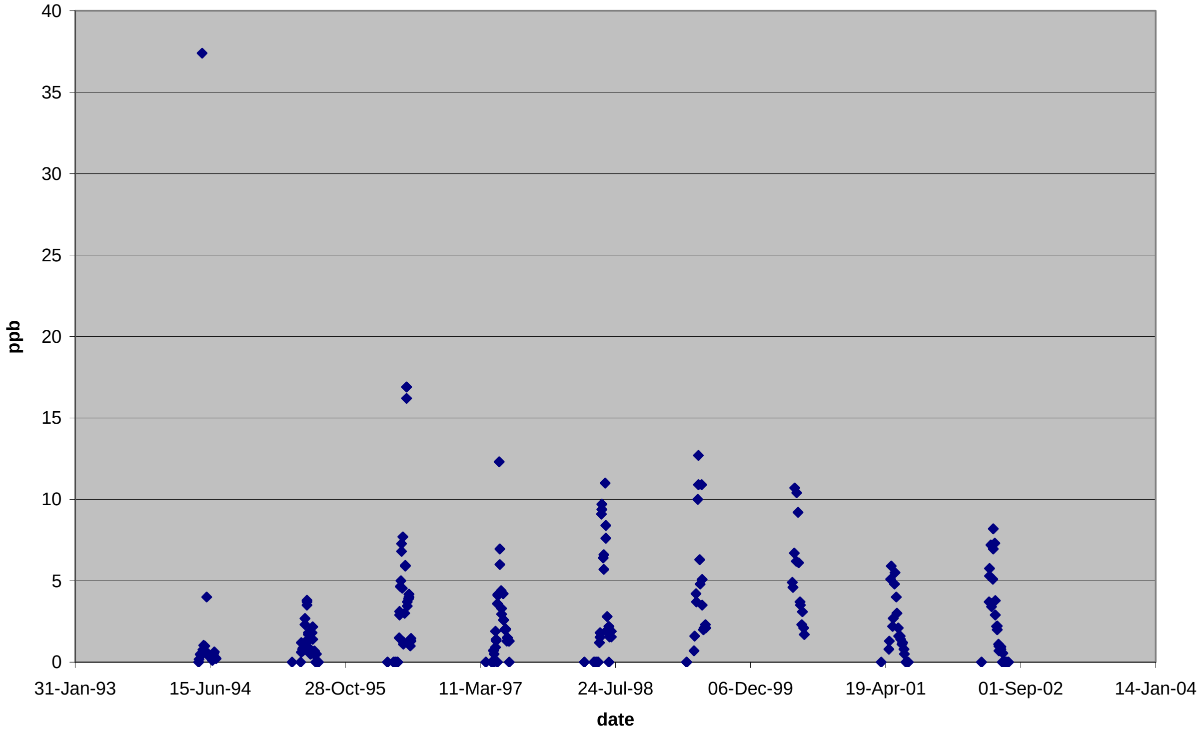

Figure 3-4. Monitoring Results Observed in the Colusa Basin Drain #5, Sacramento River, 1992 – 2002.

3.1.1.b. USGS NAWQA Surface Water Data

Data from the USGS NAWQA website for thiobencarb occurrence in surface water in California were obtained on September 21, 2009. A total of 2,117 surface water samples were analyzed for thiobencarb spanning a period from 1992 to 2009. Of these, a total of 1185 samples detected thiobencarb above the long term method detection limit of 0.002 µg/L (frequency of detection of 56%). Detections ranged from <0.002 to 4.38 µg L^{-1} (Gilliom *et al.*, 2007). The highest detections generally occurred in May, June, and July. The highest concentration was measured in a sample collected in Yolo County of the Colusa Basin in May 1997.

3.1.1.c. California Department of Pesticide Regulation (CPR) Data

CDPR maintains a database of monitoring data of pesticides in CA surface waters. Data are available from 1990-2005 for 27 counties for several pesticides and degradates. The sampled water bodies include rivers, creeks, urban streams, agricultural drains, the San Francisco Bay delta region and storm water runoff from urban areas. The database contains data from 51 different studies by federal, state and local agencies as well as groups from private industry and environmental interests. Some data reported in this database are also reported by USGS in NAWQA; therefore, there is some overlap between these two data sets. Unlike NAWQA data, the land use (*e.g.*, agriculture, urban) associated with the watershed of the sampled surface waters is not defined in the CDPR database; therefore, the available data do not allow for a link of the general use pattern and the individual data.

Surface water monitoring data were obtained from the California Department of Pesticide regulation (CDPR) and all data with analysis for thiobencarb were extracted. A total of 3,384 water samples were analyzed for thiobencarb. There were 427 detections (13% detection frequency), ranging from 0.004 and 37.4 µg/L. Detections of thiobencarb were reported in Alpine, Butte, Colusa, El Dorado, Merced, Nevada, Orange, Riverside, Sacramento, San Bernardino, San Joaquin, Stanislaus, Sutter, Yolo and Yuba counties. Samples with detections were collected between March 1991 and January 1998. The limit of quantitation ranged from 0.0002 – 1.0 µg/L.

***3.1.1.d. Halls Bayou, Arkansas (*MRID 00079986)**

A field study was conducted in rice fields bordering Halls Bayou, a tidally influenced, narrow stream that empties into West Bay near Galveston, Texas. Sampling points were located 500 feet upstream and downstream of the rice paddy and in Halls Bayou. The highest concentrations of thiobencarb were measured on a day when heavy rainfall (3.23 inches) occurred on the same day that thiobencarb was applied, resulting in an unscheduled flush overflow. Peak thiobencarb concentrations were 8.9 mg/L (8900 µg a.i./L) where the tail water exited the rice field and 690 µg/L at the point where the drainage water entered Halls Bayou. The highest concentrations measured in the Halls Bayou on days that were not associated with heavy rainfall were 83 µg/L at the upstream station (E) and 64 µg/L at the downstream station (F). These study results may not be applicable to California rice growing areas.

3.1.1.e. USGS NAWQA Groundwater Data

NAWQA Database. Ground water monitoring data from the United States Geological Survey (USGS) NAWQA program were obtained on September 21, 2009. Concentrations ranged from 0.002 to 0.025 µg/L. Thiobencarb was detected twice in Colusa County, CA, at 0.014 to 0.025 ppb (Eckel, 2008). All other detections were <0.016 - <0.002 µg/L. The highest concentration was detected in Colusa, County. The long term method detection level is 0.002 µg/L (Gilliom *et al.*, 2007).

3.1.1.f. Atmospheric Monitoring Data

Thiobencarb has been found in atmospheric samples throughout the United States, including in California and Mississippi. The maximum reported concentration of thiobencarb was 67.8 ng/m^3 Table 3-4).

Seiber *et al.* 1989. Ambient air monitoring for thiobencarb was conducted in May and June 1986 at four sites in California (Seiber *et al.*, 1989). The highest daily average concentration detected in the vicinity of current agricultural usage was 250 ng/m^3. As a control, levels of <2.0 ng/m^3 (detection limit) were monitored at a background site not near the usage areas. Several days after thiobencarb usage ceased, air concentrations were below the detection limit at the use areas. The highest measured values appeared to occur during times of usage.

Majewski *et al.* 1998. The greatest concentration of thiobencarb detected from whole air sampling on a monitoring vessel was 7.1 ng/m^3 traveling between New Orleans and Minneapolis on the Mississippi River in June 1994 (Majewski *et al.*, 1998). Thiobencarb was detected in four consecutive locations of the ten sampling points, starting south of Memphis, TN, and ending in St. Louis, MO.

Table 3-4. Summary of Air Monitoring Studies for Thiobencarb

Location	Maximum Concentration (ng/m^3)	Frequency of Detections	Limit of Det.	Date	Source
California	69.8	NR	2.0 ng/m^3	May and June 1986	(Seiber *et al.*, 1989)
Mississippi River, New Orleans to Minneapolis	7.1 0.00 (median)	40%	0.10 ng/m^3	June 1994	(Majewski *et al.*, 1998)

Abbreviations: NR= not reported; Det.=detection or reporting limit

3.1.1.g. Precipitation Monitoring Results

One study reported on thiobencarb concentrations measured in rain (Suzuki *et al.*, 2003). The highest concentration measured in rainwater from 82 samples in Eastern Japan was 0.335 µg/L (Suzuki *et al.*, 2003)(Table 3-5). The range of seasonal detection during this study was 75.0% in summer and 45.8% in winter with intermediate values for winter and spring. Values measured in Japan may not be representative of what would be found in the United States; however, these data indicate that thiobencarb may be found in precipitation and may undergo long range transport.

Table 3-5. Summary of Monitoring Studies for Thiobencarb Measuring Residues in Precipitation

Location	Median Concentration	Maximum Concentration	Frequency of Detections	Limit of Det.	Date	Source
Eastern Japan	NR	0.335 µg/L	59.8%	NR	1999-2000	(Suzuki *et al.*, 2003)

Abbreviations: NR= not reported; Det.=detection or reporting limit

3.2. Terrestrial Animal Exposure Assessment

For this assessment, spray and granular applications of thiobencarb to rice are considered. Terrestrial EECs were derived for the uses previously summarized in Table 2-5. Exposure estimates generated using T-REX are for the parent alone.

Upper-bound Kenaga nomogram values reported by T-REX are used for derivation of dietary EECs for the terrestrial phase CRLF and their potential prey. When data are absent EFED assumes a 35-day foliar dissipation half life, based on the work of Willis and McDowell (1987). The EFED RED Chapter (Mastrota, Not Specified), reported that thiobencarb residues were measured from broadleaf weeds and sedges collected 12 m downwind of the edge of the field on 0, 7, 14, and 21 days after application (Accession Number 241484; MRID 82157). The calculated foliage half-lives for broadleaf weeds and sedges were 5.4 and 8.6 days, respectively (Mastrota, 1997). These values are consistent with those estimated for other pesticides (Willis and McDowell, 1987). The upper 90th percentile confidence bound on the mean half-life value is 11.92 days and was used in this assessment.[10]

EFED included the aquatic application scenarios (rice) in the terrestrial exposure assessment. Often the treated water bodies will be quite shallow, making them accessible to terrestrial organisms. It is also likely that some thiobencarb will be deposited off the target site and onto the land adjoining the treated water bodies.

Potential direct acute and chronic effects of thiobencarb to the terrestrial-phase CRLF are initially derived by considering oral exposures modeled in T-REX for a small bird (20g) consuming small invertebrates or insects. Potential impacts to mammalian prey base were evaluated in T-REX for a small mammal (15 g) consuming short grass. Resulting dietary-based EECs (mg/kg-food) and dose-adjusted EECs (mg/kg-bw) are summarized in Table 3-6**.**

Exposure calculated as mg ai/sq ft is provided for all granular applications using the weight of one granule and the percent ai per granule as inputs into T-REX. The weight of one granule and percent ai per granule were not available for this assessment. Therefore, an LD50/ft^2 analysis was performed to evaluate potential risks to birds and mammals (for use in risk characterization) assuming a liquid formulation. The exposure used in this analysis is the mass of thiobencarb applied to a square foot area (mg/ft^2). Based on an application rate of 4.005 lbs a.i./acre (maximum single application rate), the exposure value used in the LD50/ft^2 analysis is 42 mg/ft^2.

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[10] Upper 90th percentile confidence bound on the mean = Mean + $\frac{t_{90n-1} \times \text{Standard Deviation}}{\sqrt{n}}$

Table 3-6 Summary of Dose and Dietary-based EECs Used for Estimating Dietary Risks to Terrestrial Organisms using T-REX ver. 1.4.1. for Thiobencarb Use on Rice (Liquid Formulation, ground or aerial application)

Use(s), Type of Application	App Rate (lb a.i./A, # Apps, Interval (days)	EECs for CRLF (small birds consuming small insects used as a surrogate)		EECs for Prey (small mammals 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)
Rice, ground and aerial	4, 1, n/a	540	615	960	915
Rice, ground and aerial	3, 1, n/a	405	461	720	686

Abbreviations: App= application, n/a=not applicable

T-REX is also used to calculate EECs for terrestrial insects exposed to thiobencarb. Dietary-based EECs calculated by T-REX for small and large insects (units of a.i./g) are used to bound an estimate of exposure to bees. Available acute contact toxicity data for bees exposed to thiobencarb (in units of µg a.i./bee), are converted to µg a.i./g (of bee) by multiplying by 1 bee/0.128 g. The EECs are later compared to the adjusted acute contact toxicity data for bees in order to derive RQs.

Dietary-based EECs for small and large insects reported by T-REX as well as the resulting adjusted EECs are available in Table 3-7. An example output from T-REX v. 1.3.1 is available in Appendix G.

Table 3-7. Summary EECs Used for Estimating Risk to Terrestrial Invertebrates and Indirect Effects to the CRLF using T-REX ver. 1.4.1. for Thiobencarb Use on Rice (Liquid Formulations)

Use, Method of Application	Application Rate (lbs ai/acre), # of app, App interval (days)	Small Insect	Large Insect
Rice, aerial and ground applications	4 lbs ai/A, 1 application	540	60

3.3. Terrestrial Plant Exposure Assessment

TerrPlant (Version 1.1.2) is used to calculate EECs for non-target plant species inhabiting dry and semi-aquatic areas. Parameter values for application rate, drift assumption and incorporation depth are based upon the use and related application method (Table 3-8). A runoff assessment was not included because runoff from application to rice is not expected to occur. For aerial and ground application methods, drift is assumed to be 5% and 1%, respectively. EECs relevant to terrestrial plants consider pesticide concentrations in drift and in runoff. These EECs are listed by use in Table 3-8. An example output from TerrPlant v.1.2.2 is available in Appendix H.

Table 3-8. TerrPlant Inputs and Resulting EECs for Plants Inhabiting Dry and Semi-aquatic Areas Exposed to Thiobencarb via Runoff and Drift

Use, Formulation	Application rate (lbs a.i./A)	Application method	Drift Value (%)	Spray drift EEC (lbs a.i./A)
Rice, liquid	4	aerial	5	0.20
Rice, liquid	4	ground	1	0.04
Rice, liquid	3	aerial	5	0.15
Rice, liquid	3	ground	1	0.03
Rice, granular	4.005	n/a	0	0

4. Effects Assessment

This assessment evaluates the potential for thiobencarb to directly or indirectly affect the CRLF and/or DS or modify their designated critical habitat. As previously discussed in Section 2.8, assessment endpoints for the effects determination for each assessed species include direct toxic effects on the 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 CRLF are based on toxicity information for freshwater fish (or amphibian data, when amphibian data are available), while terrestrial-phase amphibian effects are based on avian toxicity data, given that birds are generally used as a surrogate for terrestrial-phase amphibians and reptiles. Direct effects to DS are assessed based on the most sensitive E/M or freshwater fish species as DS spend time in estuarine and freshwater habitats.

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 thiobencarb, consistent with the Overview Document (USEPA, 2004). Potential direct and indirect effects to the CRLF and the DS and potential effects to critical habitat are evaluated in accordance with the methods (both screening and species-specific refinements) described in the Agency's Overview Document (USEPA, 2004).

Other sources of information, including use of the acute probit dose response relationships to establish the probability of an individual effect and reviews of the Ecological Incident Information System (EIIS), are conducted to further refine the characterization of potential ecological effects associated with exposure to thiobencarb.

A summary of the available aquatic and terrestrial organism ecotoxicity information and the incident information for thiobencarb are provided in the following sections. A summary of the available data directly used in this assessment is presented. A more comprehensive list of the available toxicity data is included in Appendix I.

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, 2009). Open literature data presented in this assessment were obtained from a recent assessment (Davy, 2008), from a recent HED toxicology assessment (Lewis, 1997) as well as ECOTOX information obtained on February 28, 2009. In order to be included in the acceptable ECOTOX data summary, 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 'target' terrestrial plant species, which include efficacy studies, are not currently considered in deriving the most sensitive endpoint for terrestrial plants. 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 data are not included in the ECOTOX open literature summary table provided in Appendix K. The list of citations including toxicological and/or efficacy data on target plant species not considered in this assessment is provided in Appendix J.

Meeting the minimum criteria for inclusion in ECOTOX does not necessarily mean that the data are suitable for use in risk estimation. 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. In general, effects data in the open literature that are more conservative than the registrant-submitted data are considered. The degree to which open literature data are quantitatively or qualitatively characterized for the 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 to define the action area for thiobencarb.

Citations of all open literature 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 Appendix J. Appendix J 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 K. Appendix L is a summary of the human health effects data for thiobencarb.

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 the Ecological Incident Information System (EIIS), are conducted to further refine the characterization of potential ecological effects associated with exposure to thiobencarb. A summary of the available aquatic and terrestrial ecotoxicity information and the incident information for thiobencarb are provided in Sections 4.1 through 4.5, respectively.

4.2. Toxicity Categories

Toxicity to fish, aquatic invertebrates, birds, and mammals is categorized using the system shown in Table 4-1 (USEPA, 2004). For non-target terrestrial insects, chemicals with LD_{50} values of <2, 2 – 11, and >11 µg/bee are classified as highly toxic, moderately toxic, and practically nontoxic, respectively. Toxicity categories for terrestrial and aquatic plants have not been defined.

Table 4-1. Categories of Acute Toxicity for Terrestrial and Aquatic Animals.

Toxicity Category	Aquatic Animals [LC_{50}/EC_{50} (µg/L)]	Birds and Mammals [LD_{50} (mg/kg-bw)]	Birds [LC_{50} (mg/kg-diet)]
Very highly toxic	< 100	<10	<50
Highly toxic	100 – 1,000	10 – 50	50 – 500
Moderately toxic	> 1,000 – 10,000	51 – 500	501 – 1000
Slightly toxic	> 10,000 – 100,000	501 – 2000	1001 – 5000
Practically nontoxic	> 100,000	>2000	>5000

4.3. Toxicity of Thiobencarb to Aquatic Organisms

Table 4-2 summarizes the most sensitive aquatic toxicity endpoints, based on an evaluation of both the submitted studies and the open literature, as previously discussed. A brief summary of submitted and open literature data considered relevant to this ecological risk assessment for the CRLF and DS is presented below. Additional information is provided in Appendix I. Values used in risk quotient calculations are shown with an asterisk. All endpoints are expressed in terms of the active ingredient (a.i.) unless otherwise specified. All available acute aquatic toxicity endpoints ranged from 17 – 47,820 µg/L (Appendix I). Limited data available on amphibians suggest that they are not one of the more sensitive species (96-hr LC_{50} ranged from 1.3 – 6.5 mg/L; see Section 4.3.1.d); however, these data are highly uncertain.

Table 4-2. Aquatic Toxicity Profile for Thiobencarb

Assessment Endpoint	Acute/ Chronic	Species TGAI/TEP % ai	Toxicity Value	Citation or MRID #	Comment
Freshwater fish (surrogate for aquatic-phase amphibians)	Acute	Bluegill sunfish *Lepomis macrochirus* TEP 10%	96-hr LC_{50} = 560 µg/L*	MRID 00050665	Acceptable, previous discrepancies are resolved through updated 850.1075 guidelines (*i.e.* fish are <0.5g but current methods require fish <3g without a lower limit)
	Acute	Striped Bass *Morone saxatilis* TEP 85.2%	96-hr LC_{50} = 440 µg/L*	E15472[2] (Fujimura *et al.*, 1991)	Supplemental-quantitative. The study did not have a solvent control; however, later studies at the same laboratories including solvent controls indicated that the solvent control was not significantly different from the negative control.
	Chronic	Striped Bass *Morone saxatilis* TEP 85.2%	NOAEC/ LOAEC = 21/ 23 µg/L* (9-day posthatch, survival)		Quantitative. Hatching success was not evaluated and this may result in an underestimated risk to hatching success.
Freshwater invertebrates	Acute	*Daphnia magna* TGAI 94.4%	48-hr EC_{50} = 101.2 µg/L*	MRID 00025788	Acceptable
	Chronic	*Daphnia magna* TGAI ~95%	21-day NOAEC/LOAEC = 1.0/ 3.0 µg ai/L* (offspring produced)	MRID 00079098	Acceptable.
Estuarine/ marine fish	Acute	Atlantic silverside *Menidia menidia* TGAI 90%	96-h LC_{50} = 204 µg/L*	E11868[2] (Borthwick *et al.*, 1985)	Supplemental- quantitative
	Chronic	Atlantic silverside	NOAEC = 6 µg a.i./L, estimated*	n/a	Estimated based on ACR for the striped bass of 770/21 = 37 and the Atlantic silverside 96-h LC_{50} of 204 µg/L.
Estuarine/ marine invertebrates	Acute	Mysid *Americamysis bahia* TGAI 94.6%	96-hr LC_{50} = 150 (110-200) µg ai/L*	MRID 00050667	Acceptable. Mysid were slightly older than suggested. Nominal
	Chronic	Opossum Shrimp *Neomysis mercedis*	NOAEC/ LOAEC = 3.2/6.2 µg ai/L* (survival of offspring)	MRID 43976801/E Supplemental, Quantitative (USEPA,	Open literature study (MRID 43976801) that is referenced as supplemental in the RED. Data were submitted by the same author under MRID

Assessment Endpoint	Acute/ Chronic	Species TGAI/TEP % ai	Toxicity Value	Citation or MRID #	Comment
				1997)	40651314. No DER was completed, an open literature review has been completed. Gravid females were replaced if they died for the first fourteen days. Replacement was not reported.
Aquatic plants	N/A	Green algae *Selenastrum capricornutum*	5-day EC_{50} = 17 (12 – 26) µg ai/L* NOAEC = 13 µg ai/L	MRID 41690901	Acceptable. Endpoint based on decreased cell density.
	N/A	Green algae *Scenedesmus acutus*	96-hr EC_{50} = 17 (16 – 19) µg ai/L* NOAEC = 5 µg ai/L	E17114[2] (Sabater and Carrasco, 1996)	Supplemental-qualitative. Endpoint based on percent inhibition of growth. LOAEC = 9 µg ai/L
	N/A	duckweed *Lemna gibba*	14-day EC50 = 770 (380 – 1600) µg ai/L* NOAEC = 14 µg ai/L	MRID 41690901	Acceptable. Endpoint based on decreased frond production.

Abbreviations: a.i.=active ingredient; C.I. = confidence interval; NOAEC = No observed effect concentration; NOAEL = No observed adverse effect level; LC_{50} = Lethal concentration to 50% of the test population; EC_{50} = Effect concentration to 50% of the test population; IC_{50}= inhibition concentration resulting in a 50% inhibition in the test population response (*e.g*., growth); TGAI=technical grade active ingredient; TEP=typical end-use product; DO = dissolved oxygen; ACR=acute-to-chronic ratio

1-Other endpoints affected include F_0 maturation (week 24) wet weight (Males), F_0 maturation (week 37) wet weight (Females and Males), F_0 eggs/female, F_0 eggs/spawn, F_1 hatching success, F_1 4-week survival, F_1 4-week length).

2-ECOTOX references are designated with an E followed by the ECOTOX reference number.

4.3.1. Toxicity to Freshwater Fish and Aquatic-Phase Amphibians

Fish toxicity data were used to evaluate potential direct effects to aquatic-phase CRLF and the DS and indirect effects to the CRLF. A summary of acute and chronic fish and aquatic-phase amphibian data, including data from the open literature, is provided in the following sections.

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

Thiobencarb and thiobencarb formulations are moderately to highly toxic to freshwater fish. The only studies conducted using the technical grade active ingredient (TGAI) or typical end-use product (TEP) with greater than 95% a.i. were considered supplemental and/or qualitative and had incomplete reporting of the test procedures. The supplemental results of these studies indicated that the 96-hr LC_{50} values for carp, bluegill sunfish, channel catfish, and rainbow trout ranged from 1180 - 2800 µg a.i./L (MRIDs 00080859, 00080851; see Appendix I). The only fully acceptable submitted studies on the acute toxicity of thiobencarb to fish were conducted with Bolero 10 G. The 96-hour LC_{50} for bluegill sunfish exposed to Bolero 10 G (10% a.i.) was

560 µg a.i./L (MRID 00050665). This result is inconsistent with the results of other acute tests for the bluegill sunfish where LC_{50} values for TEPs ranged from 1660 – 2800 µg a.i./L (MRIDs 00080851, 139051). Results of tests with the TEP or TGAI with less than 91% a.i. ranged from 260 – 2480 µg a.i./L indicating that thiobencarb formulations are highly toxic to moderately toxic to freshwater fish (MRIDs 40651315, 00080851). The most sensitive species was the white sturgeon[11] and this value may only be used qualitatively due to incomplete information available in the test report. As little information is available on the studies conducted on the TGAI, it is not possible to determine whether the TEP or TGAI is more toxic. The TGAI studies are not as reliable as the TEP studies because raw data were not available and there was incomplete reporting of the test results. Therefore, results from the TEP will be used quantitatively in the risk assessment and it will be assumed that the toxicity to the formulation and TGAI are similar. This assumption is supported from open literature studies (Harrington, 1990).

In addition to the acute toxicity studies discussed above, several acute toxicity studies with freshwater fish were identified from the open literature (see Appendix I) which report more sensitive values. The lowest of these more sensitive acute LC_{50} values is 260 µg a.i./L (Bailey, 1984) for the white sturgeon, *Acipenser transmontanus*[12]. This study was evaluated and considered qualitative as there was no solvent control in the study, the percent active ingredient in the test material was not reported, and not enough information was available on the preparation of test solutions. This value is considered qualitative. Additionally, several toxicity studies were conducted examining toxicity to the striped bass (Fujimura *et al.*, 1991). The studies examining toxicity to the striped bass had some endpoints lower than those submitted and some of those endpoints did not include solvent controls. These 96-hour LC_{50} values ranged from 430 – 550 µg/L. Control mortality was greater than 10% in the test with the lowest endpoint of 430µg/L; thus, this value may only be used qualitatively. Solvent controls were completed for the studies on striped bass completed in 1989 and no differences were observed between the negative and solvent controls. Therefore, it may be assumed that the solvents used did not influence the results and the 96-hr LC_{50} of 440µg/L may be used quantitatively. The other endpoints reported in the study are also considered quantitative and may be used to calculate an acute-to-chronic ratio and/or risk quotient.

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

One supplemental life-cycle study on fathead minnows (*Pimephales promelas*) was available to the Agency to evaluate the effects of chronic exposure to thiobencarb to freshwater fish (MRID 45695101). The study demonstrated that exposure to thiobencarb at 110 µg a.i./L has the potential to cause reproductive toxicity. At this concentration the following endpoints were statistically different from controls: F_0 survival at 4- and 8-weeks, F_0 growth at, 8 weeks (weight and length), 24 weeks (♂ and ♀ weight and length), and 37 (♂ and ♀ weight and length) weeks post-hatch; F_0 reproduction (eggs/female, spawns/female and eggs/spawn); F_1 hatching success

[11] The open literature white sturgeon study was submitted to EPA and is also available in the open literature. It is discussed further in the open literature discussion.
[12] A study report describing this study was also submitted to the Agency.

(58%); F_1 survival at 4 weeks post-hatch; F_1 time-to-hatch; and F_1 growth (weight and length) at 4 weeks post-hatch. The NOAEC and LOAEC were 0.53 and 110 µg a.i./L, respectively. The study only had two replicates. The low number of replicates and variability in the endpoints may have resulted in an inability to statistically detect differences between treatments and controls and thus may overestimate toxicity endpoints. Observed results suggested that if more replicates were available a difference may have been statistically significant between the control and 53 µg a.i./L treatment group.

Open literature data are available for the Chinook salmon and Striped bass, with the most sensitive endpoint reported for the Striped bass (E015472). The NOAEC and LOAEC from that study for survival was 21 and 36 µg/L, respectively. The test was started using 1-hour prehatch striped bass. A NOAEC was not determined in an early life stage study with striped bass at 8-day post hatch at the test initiation. At the lowest measured test concentration of 23µg/L, survival was 63% as compared to 85% in controls. The NOAEC of 21µg/L was used to evaluate chronic risk the CRLF and DS in freshwater.

4.3.1.c. Freshwater Fish: Sublethal Effects and Additional Open Literature Information

No additional acceptable studies from the open literature were identified for freshwater fish that: established more sensitive acute or chronic endpoints than the data listed above; filled critical data gaps; presented a toxicity profile for under-represented taxa (*e.g*., toxicity data for amphibians); or provided information on sub-lethal effects that could be clearly and reasonably linked to relevant assessment endpoints (*i.e*., survival, reproduction, and growth) at concentrations lower than the most sensitive endpoints used to quantitatively evaluate risk.

4.3.1.d. Aquatic-phase Amphibian: Acute and Chronic Studies

Acute, static-renewal, 96-hour early, middle, and late stage amphibian larvae toxicity tests were completed on six species of amphibians examining toxicity of thiobencarb (TGAI, 99% thiobencarb), an emulsifiable concentrate (EC, 50% thiobencarb), and three granular formulations containing thiobencarb and other pesticide active ingredients (Saka, 1999). The results indicated that the formulation toxicity and toxicity of the TGAI were similar. Additionally, the 96-hr LC_{50} values for amphibians were near 1.3 – 6.5 mg a.i./L, suggesting that amphibians may be less sensitive to thiobencarb than fish. These data are highly uncertain and may only be used qualitatively (see Appendix I for more information on this study).

4.3.2. Toxicity to Freshwater Invertebrates

A summary of acute and chronic freshwater invertebrate data, including data published in the open literature, is provided below in Sections 4.1.2.1 through 4.1.2.3.

4.3.2.a. Freshwater Invertebrates: Acute Exposure Studies

The results indicate that thiobencarb is moderately to highly toxic to aquatic invertebrates on an acute basis with LC_{50} values ranging from 101.2 to 6500 µg a.i./L. (see Appendix I). No open literature values resulted in more sensitive acute values for freshwater invertebrates. Available data indicate that invertebrates have either similar to or less sensitivity to the TEP as compared to the TGAI. The 48-hr LC_{50} for the water flea is 101.2 µg a.i./L (95% Confidence Interval, C.I., is 73.8 – 138.7 µg a.i./L) and 210.7 µg a.i./L (95% C.I. is 175.7 - 252.7 µg a.i./L) for the TGAI and TEP, respectively (MRIDs 0025788, 00079118). The endpoint for the TGAI is the lowest available value and will be used in the risk assessment.

4.3.2.b. Freshwater Invertebrates: Chronic Exposure Studies

Toxicity of thiobencarb (95.2 - 95.9% a.i.) was examined in a life-cycle toxicity study for daphnid, *Daphnia magna* (MRID 00079098). The NOAEC and LOAEC were 1.0 µg a.i./L and 3.0 µg a.i./L, respectively. Chronic effects observed were reduced number of young produced and adult mortality. These results indicate that concentrations of thiobencarb greater than 1 µg a.i./L can be detrimental to the survival and reproduction of freshwater invertebrates. Additionally, a supplemental 28-day sediment toxicity test on the *Chironomus Riparius* showed that sediment toxicity (decreased percent emergence and altered sex ratio) occurred when the time weighted average concentration of thiobencarb in overlying water was 420 µg/L (MRID 46091402). The NOAEC in the study was 180 µg/L.

4.3.2.c. Freshwater Invertebrates: Open Literature Data

No open literatures studies reported lower median lethal concentrations than those obtained from submitted studies for freshwater invertebrates.

The most important food organism for all sizes of the Delta smelt has been reported to be the copepod *Eurytemora affinis* (USFWS, 1995, 2004a), which is a marine copepod. No E/M studies were submitted examining toxicity to copepods. Supplemental toxicity data are available from the open literature for a non-native freshwater *cyclopoida* and *calanoida* (ECOTOX E062293). In this field study, conducted with a formulation of thiobencarb, the 7-day LOAEL was 0.1875 mg a.i./L (effects observed were emergence success).

4.3.3. Toxicity to Estuarine/Marine Fish

A summary of acute and chronic E/M fish data, including data published in the open literature is provided below in Sections 4.1.3.1 through 4.1.3.3.

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

Thiobencarb and thiobencarb formulations are moderately to highly toxic to E/M fish on an acute basis. Sheepshead minnow were more sensitive to the TGAI than the TEP. The 96-hr LC_{50} for the TGAI was 660 (95% CI = 600-800) and 900 (95% CI = 700-1200) µg a.i./L while the value for the TEP was 96-hr LC_{50} was 1400 (95% CI = 1100-1800) µg a.i./L (MRID 00079112, 00079110, 00079111). The lowest 96-hr LC_{50} from a submitted study for the TGAI is 660 µg a.i./L (MRID 00079112). Several open literatures studies were available in the open literature with the majority of the values being below the lowest submitted value.

All of the open literature acute values from EPA's ECOTOX database for E/M fish are presented in Appendix I. There were 25 acute LC_{50} values from two open literature studies representing four E/M fish species. Almost all of these values were below the E/M fish studies provided by the registrant (660 µg a.i./L). The lowest of these more sensitive acute 96-hr LC_{50} values is 204 µg a.i./L for the Atlantic silverside, *Menidia menidia* (Borthwick *et al.*, 1985). This study was evaluated and considered supplemental quantitative. Endpoints measured for California grunion, Atlantic silverside, and Tidewater silverside ranged from 204 – 1174 µg a.i./L (ECOTOX Number E11868).

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

One study examining toxicity of thiobencarb to the Sheepshead minnow is available. The NOAEC measured in an early life stage study for 28-day post hatch Sheepshead minnow was <150 µg/L and effects on wet weight were observed at all treatment levels (MRID 00079112). Additionally, reduced survival and hatching success were observed at 230 and 600 µg a.i./L. No chronic E/M data were submitted to the Agency or available in the acceptable ECOTOX report. A chronic value was estimated using the an acute-to-chronic (ACR) ratio calculated for freshwater fish and the lowest 96-hr LC_{50} of 204 µg a.i./L for the Atlantic Silverside (Fujimura *et al.*, 1991). The estimated chronic NOAEC value for the Atlantic Silverside was calculated as follows:

$$\frac{\text{Striped Bass Highest 96-hr } LC_{50}}{\text{Striped Bass NOAEC}} = \frac{\text{Atlantic Silverside 96-hr } LC_{50}}{\text{Atlantic Silverside NOAEC}}$$

Where:

the acute striped bass value is based on the highest quantitative 96-hr LC_{50} values of 770 µg a.i./L (Fujimura *et al.*, 1991),

the chronic striped bass value is based on the NOAEC of 21 µg a.i./L from the same study and,

the acute Atlantic silverside 96-hr LC_{50} is 204 µg a.i./L as described previously (Fujimura *et al.*, 1991).

Therefore, 770/21 = 204/X

and,

Estimated Atlantic silverside NOAEC = (204 x 21)/770 = 6 µg a.i./L.

This estimated NOAEC of 6 µg a.i./L for Atlantic silverside will be used to quantitatively estimate chronic risk of thiobencarb to DS. Risk quotients could underestimate risk as the hatching success was not measured for the NOAEC used to calculate the ACR.

4.3.4. Toxicity to Estuarine/Marine Invertebrates

A summary of acute and chronic E/M invertebrate data, including data published in the open literature, is provided below in Sections 4.1.4.1 through 4.1.4.3.

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

Estuarine/marine aquatic invertebrate toxicity data are used to evaluate potential indirect effects to the DS because they depend on aquatic invertebrates for food. For the indirect effects assessment, the most sensitive aquatic invertebrate species is initially used for risk estimation, which is consistent with USEPA (2004). The most sensitive E/M aquatic invertebrate tested is the mysid shrimp (*Mysidopsis bahia*) (96-hr LC_{50} = 150, 95% C.I. = 110-200 µg a.i./L) (MRID 00050667). Other E/M invertebrates have similar to less sensitivity to thiobencarb when compared to the mysid shrimp. All 96-hr LC_{50} values for all shrimp and Eastern oyster species ranged from 150 – 1100 µg a.i./L (see Appendix I). The fiddler crab was less sensitive than other species with a 96-hr LC_{50} value of 4400 µg/L (MRID 00079113).

One open literature study resulted in a lower endpoint than those observed in submitted studies (E090259). The endpoint is not useable in the risk assessment due to incomplete reporting of test procedures and other major limitations of the study. All other acute endpoints in the acceptable ECOTOX report were higher than 150 µg/L.

The most important food organism for all sizes of the Delta smelt has been reported to be the copepod *Eurytemora affinis* (USFWS, 1995, 2004a), which is a marine copepod. No E/M studies were submitted examining toxicity of thiobencarb to E/M copepods.

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

Three studies are available examining chronic toxicity to E/M invertebrates. Open literature toxicity data from chronic exposure to thiobencarb are available for three the Mysid, Opossum, and Grass shrimp. The Opossum shrimp (*Neomysis mercedis*) study, conducted with technical grade thiobencarb, resulted in the lowest NOAEC of 3.2 µg a.i./L based on reduced survival of offspring at 6.2 µg a.i./L (MRID 43976801 or 40651314). No open literatures studies with lower endpoints were found in the acceptable ECOTOX report.

4.3.5. Toxicity to Aquatic Plants

Aquatic plant toxicity studies are used as one of the measures of effect to evaluate whether thiobencarb may affect primary production. Aquatic plants may also serve as dietary items of aquatic-phase CRLFs. In addition, freshwater vascular and non-vascular plant data are used to evaluate a number of the PCEs associated with the critical habitat impact analysis.

Thiobencarb is toxic to the freshwater green alga (*Pseudokirchneriella subcapitata*, formerly *Selenastrum capricornutum)*, with a 120-hr EC_{50} of 17 µg a.i./L and a NOAEC of 13 µg a.i./L, based on reduced cell density (MRID 41690901). The marine diatom was also sensitive to thiobencarb with a 120-hr EC_{50} of 73 µg a.i./L and NOAEC of 18 µg a.i./L (MRID 41690901). The freshwater diatom, and blue-green algae were less sensitive with 120-hr EC_{50} values of 380 and >3100 µg a.i./L (MRID 41690901). The aquatic vascular plant tested, duckweed (*Lemna gibba*), is less sensitive to thiobencarb than the freshwater green alga [*i.e.*, 14-day EC_{50} = 770 µg a.i./L; NOAEC = 140 µg a.i./L; based on decreased frond production] (MRID 41690901).

Two open literature studies are available with endpoints lower than those measured in submitted studies. A 4-day NOAEC was reported for green algae (*Scenedesmus acutus*) of 5 µg a.i./L, with effects on percent inhibition of growth observed at 2 µg a.i./L (E15718). The LC_{50} measured in the study was the same as that measured in the submitted study (see Table 4-2).

4.3.6. Aquatic Field/Mesocosm Studies

The conclusion of high risk to aquatic organisms, based on results from laboratory toxicity tests, triggered the requirement for aquatic field testing with thiobencarb (GLN 72-7). The following aquatic field studies have been conducted on the use of thiobencarb on rice.

Table 4-3. Summary of submitted aquatic field studies on the use of thiobencarb on rice

Title	Location and Date	Reference	Performed By	Sponsor	Classification
Impact of Bolero Runoff on a Brackish Water Ecosystem	Matagorda, Texas 1982 – 1984	MRIDs 42130705 & 42130708	Fujie, 1983.	Chevron Chemical Company	Acceptable[1]
Thiobencarb: Studies on Residue Level and Behavior in Selected Irrigation Creeks in Agricultural Areas in Saga Prefecture, Southwestern Japan	Saga Prefecture, Kyushu, Japan 1975	MRID 00028183	Ishikawa, 1975	Unknown	Supplemental

[1] Following the review of this study, an additional aquatic field study was requested to monitor aquatic residues in other localities where rice is grown. This additional study, however, was waived in December 1993.

4.3.6.a. Matagorda Study

A large aquatic field study was conducted in 1982-1984 near Matagorda, Texas. The site consisted of a rice field that drained through a ditch into the tidal waters of the lower Colorado River of eastern Texas. This estuarine area is a complex and highly important ecosystem that supports many commercial species. No thiobencarb applications were made in 1982; this year provided an estimate of background levels of thiobencarb. Thiobencarb concentrations were as high as 9 µg a.i./L in 1982. In 1983 and 1984, approximately 500 acres of the field were treated with thiobencarb at a rate of 4 lbs a.i. per acre. Fields were flushed with water within 3 to 12 days after application. Data collected from 1982 through 1984 included (1) residues of thiobencarb in water, sediment, fish and shrimp; (2) catch per unit effort measurements of fish and aquatic invertebrates; and (3) percentages of grass shrimp (*Palaemonetes pugio*) that were gravid. While samples were collected during all three years of the study, the sampling effort on the third year was very poor.

A control station was also planned on the Colorado River upstream of the confluence with the drainage ditch. However, during the course of the study, the Agency and the registrants agreed that this station could not serve as a control for the field study because it contained preexisting residues of thiobencarb. It was therefore only possible to compare residues and biological samples collected during 1983 and 1984 to those collected during 1982, before the initial treatment. This represents a shortcoming of this study since the results could have been influenced by yearly fluctuations in environmental conditions that are unrelated to the applications of thiobencarb. Another shortcoming is that other pesticides (ordram, basegran, machette, and propanil) were applied to fields that drain into the test ditch during the period of this study. The toxicity of these pesticides could have contributed to the observed effects.

The results of the study were:

1. Residues of thiobencarb were transported into the estuary via runoff and drift. Maximum residues measured in water, sediment, fish, and shrimp were 25.1 µg/L, 50 µg/L, 2400 µg/L, and 970 µg/L, respectively.

2. Although the overall population of fish was apparently not affected, marked declines were observed during the treatment years in three species, *Gambusia affinis*, *Dormitator maculatus*, and *Poecilia latipenna*.

3. Several taxa of aquatic invertebrates showed substantial decline in numbers caught per unit effort. Species richness and diversity also declined significantly during treatment years.

4. The percentage of gravid shrimp decreased significantly in 1983 compared to 1982. The decline was about 50% at stations 1 and 2, and averaged 23% for all four stations. Sampling was inadequate to assess the effect on the percentage of gravid shrimp in 1984.

5. A kill of the fish menhaden (*Brevoortia patronus*) was observed in the area where the field runoff entered the drainage ditch. It occurred at the point of discharge from the drainage

canal, one to two days after a post-application flush of the rice fields. Although other pesticides that were applied that year (ordram, basegran, and propanil) may have been present in the tailwater, this kill was attributed to thiobencarb contamination because the dead fish contained high residues of thiobencarb (mean of 3.56 ppm).

6. Field BCF for thiobencarb was estimated to be 109X for fish and 44X for shrimp.

Declines in fish, aquatic invertebrates, and gravid shrimp cannot conclusively be attributed to the use of thiobencarb. Nevertheless, the findings in the field were consistent with effects demonstrated in laboratory studies. They suggest that the application of thiobencarb to rice fields may result in significant environmental damage to the adjacent estuarine habitat in Texas. Possible effects include chronic effects to sensitive fish, acute and chronic effects to ecologically important aquatic invertebrates, chronic effects to grass shrimp and possibly to commercial shrimp, and indirect detrimental effects to organisms at higher trophic levels that depend on these organisms for food.

4.3.6.b. Japan Study

The EFED reviewed a study that measured residues of thiobencarb in creek water after application to rice paddies in Japan. Thiobencarb was applied in the form of 7% granules at a rate of 30 kg/ha, which is equivalent to 1.9 lb a.i./A. Water samples were taken from ten stations along creeks that flow through the rice fields and drain into the Hayatsue River. Water sampling was conducted from March through November, with thiobencarb treatments being made from June 28 through July 2. The creeks served as storage for irrigation water until May, when the water is pumped onto the fields. The creeks resembled large ponds during the storage period.

Very low thiobencarb concentrations (0.2 µg a.i./L or less) were reported at all stations in March and April before applications were made. Concentrations peaked at the sampling period of July 1, when concentrations at most stations were between 20 and 40 µg a.i./L. The greatest concentration was measured was 40.5 µg a.i./L. Concentrations declined fairly rapidly thereafter; the half-life of thiobencarb in creek water was estimated to be 8.8 days. This rate of decline represents dilution as well as biological and physical degradation processes. EFED cannot interpret the significance of these results or extrapolate conclusions to other areas because of the lack of important information on the test conditions, such as flow rates within the creeks and rainfall during the study.

4.3.6.c. Conclusions and Uncertainties with Field Studies

A difficulty with the field studies was that water flow measurements were not made, making it impossible to discern effects of dissipation versus dilution. While water residues were generally short-lived, it is not clear whether thiobencarb residues were broken down by chemical or biological forces, or they were swept away and diluted by tidal flow. Because it is possible that dilution was the primary mode of dissipation in all three studies, the rate at which thiobencarb degrades by chemical or biological means in estuaries remains unknown. Thiobencarb residues thus may persist longer in other areas where dilution is of less importance in the dissipation of residues.

The two biological field studies demonstrate that application of thiobencarb on rice can cause significant contamination to water, sediments, and aquatic organisms in off-site aquatic habitats. Harm to estuarine and freshwater ecosystems is possible when thiobencarb is used in southeastern United States. Although shortcomings of these studies make it impossible to identify thiobencarb as the sole cause of observed adverse effects, the studies fail to refute the Agency's presumption that the use of thiobencarb on rice results in severe effects on aquatic ecosystems.

4.4. Toxicity of Thiobencarb to Terrestrial Organisms

Table 4-4 summarizes the most sensitive terrestrial toxicity endpoints, based on an evaluation of both the 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 I.

Table 4-4. Terrestrial Toxicity Profile for Thiobencarb

Endpoint	Acute/ Chronic	Species	Toxicity Value Used in Risk Assessment	Citation MRID/ ECOTOX reference No.	Comment
Birds (surrogate for terrestrial-phase amphibians)	Acute	Bobwhite quail (*Colinus virginianus*) TGAI 96.9%	Acute oral LD_{50} >1938 mg/kg-bw	MRID 42600201	Acceptable
	Acute	Mallard duck *Anas platyrhynchos*	5-day LC_{50} >5000 mg a.i./kg-diet	MRID 44846206	Acceptable No treatment related mortality. NOAEC = 648 mg a.i./kg-bw (body weight gain)
	Chronic	Mallard duck *Anas platyrhynchos* TGAI 95.5%	1-generation NOAEC/ LOAEC = 100/ 300 mg a.i./kg-diet	MRID 00025778	Supplemental. Raw data were not complete for hatching success and 14-day survivor weight. The effects observed at the LOAEC were decreased eggs laid and hatchlings per live 3 week embryo.
Mammals	Acute	Rat – male TGAI 96%	Acute oral LD_{50} = 1033 mg/kg-bw	MRID 42130701	Acceptable
	Chronic	Fischer 344 Rat TGAI 95.3%	2-generation NOAEL/ LOAEL= 20/ 100 mg/kg/day or 1/ 5 mg/kg-bw	MRID 40446201	Acceptable. The effects observed were decreased bw gain, food consumption, and food efficiency. No reproductive effects were observed at any test level and the reproductive NOEAL ≥

Endpoint	Acute/ Chronic	Species	Toxicity Value Used in Risk Assessment	Citation MRID/ ECOTOX reference No.	Comment
					100 mg/kg/day.
Terrestrial invertebrates	Acute Contact	Honey Bee *Apis Mellifera* TGAI 97.2%	Acute contact 48-hr LD_{50}> 100 µg/bee	MRID 46059804	15 percent mortality observed at 100 µg/bee, the highest dose tested.
Terrestrial plants	N/A	Seedling Emergence Monocots	EC_{25}/ NOAEC = 0.019/ 0.0051 lb a.i./A	MRID 41690902	Most sensitive species was ryegrass. The most sensitive endpoint was mortality.
	N/A	Seedling Emergence Dicots	EC_{25}/ NOAEC = 0.082/ 0.071 lb a.i./A	MRID 41690902	Most sensitive species was cabbage. The most sensitive endpoint was shoot length.
	N/A	Vegetative Vigor Monocots	EC_{25}/ NOAEC = 0.073/ 0.020 lb a.i./A	MRID 41690902	Most sensitive species was ryegrass. The most sensitive endpoint was shoot length.
	N/A	Vegetative Vigor Dicots	EC_{25}/ NOAEC = ND/ <0.12 lb a.i./A EC25/NOAEC = 1.2/ 0.80 lb a.i./A* (soybean)	MRID 41690902	Most sensitive species was cucumber. Shoot weight and root weight were the most sensitive endpoints. For soybean, the most sensitive endpoint was root weight.

N/A: not applicable; ND = not determined; bw=body weight; hr=hour

4.4.1. Toxicity to Birds and Terrestrial-Phase Amphibians

As specified in the Overview Document, the Agency uses birds as a surrogate for terrestrial-phase amphibians when toxicity data for each specific taxon are not available (USEPA, 2004). A summary of acute and chronic bird data, including data published in the open literature is provided below in Sections 4.2.1.1 through 4.2.1.4. No terrestrial phase amphibian data were available in the ECOTOX summary of acceptable ecotoxicity data.

4.4.1.a. Birds: Acute Exposure (Mortality) Studies

Only two acute avian toxicity studies were available. An acute oral toxicity study on the bobwhite quail indicates that thiobencarb is slightly toxic to practically nontoxic to bobwhite quail (Acute oral LD_{50} >1938 mg a.i./kg-bw, MRID 42600201). No mortality occurred at the highest dose tested and the only sublethal effect observed was a slight decrease in body weight during the first three days after dosing. No mortality was observed in a subacute dietary study with the mallard duck with the highest dose tested being 5000 mg a.i./kg-bw (MRID 44846206). Sublethal effects observed in treatments of 1080 mg a.i./kg-diet and above include: increased water consumption, lack of coordination, smaller appearance, decreased body weight gain (57-

10% in treatments versus 76 - 77% in control), and decreased food consumption. Birds completely recovered after 28 days.

Acute oral toxicity studies with a passerine and water fowl species were not available. The limited data available on birds could result in an underestimation of toxicity.

No data were available in the accepted ECOTOX data examining acute toxicity of thiobencarb to birds.

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

An avian reproduction study with the mallard duck (MRID 00025778) resulted in a NOAEC of 100 mg a.i./kg-diet, based on decreased eggs laid and hatchlings per live three week embryo. Sublethal effects observed in the study at 30 mg a.i./kg-diet and above included depression of coordination, lower limb weakness, prostate posture, loss of reflexes, and wing droop. These effects disappeared after two days. In another supplemental study with the mallard duck, the 14 day survivor weight was significantly less at 231 mg a.i./kg-diet (NOEAC = 115 mg a.i./kg-diet) and normal hatching eggs laid and eggs set were decreased at 338 mg a.i./kg-diet (MRID 45140601). Reproductive and growth endpoints were also affected in quail. In Japanese quail, the fertility and hatchability were reduced at 1000 mg a.i./kg-diet (MRID 00080848). In the bobwhite quail, the percent normal embryos and weight of hatchlings was reduced at 930 mg a.i./kg-diet (MRID 43075401).

4.4.1.c. Terrestrial-phase Amphibians: Acute and Chronic Studies

No data are currently available for the effects of thiobencarb on terrestrial-phase amphibians.

4.4.2. Toxicity to Mammals

A summary of acute and chronic mammalian data, including data published in the open literature, is provided below in Sections 4.2.2.1 through 4.2.1.2. A more complete analysis of toxicity data to mammals is available in Appendix L, which is a copy of the HED chapter prepared in support of the reregistration eligibility decision completed in 1997.

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

Thiobencarb is slightly toxic to mammals on an acute oral exposure basis. An acute oral study with rats (*Rattus sp.*) resulted in an LD_{50} value of 1033 mg a.i./kg-bw (MRID 42130701).

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

In a combined chronic toxicity/carcinogenicity feeding study (MRID 00154506), Fischer 344 rats received 0, 20, 100 or 500 mg/kg-diet (approximately 0, 1, 5, and 25 mg/kg/day by standard conversion methods) technical thiobencarb (95.3% a.i.) in the diet for two years. Systemic toxicity was noted at 100 mg/kg-diet and above as decreased body weight gain, food consumption and food efficiency. There was also an increase in blood urea nitrogen. No evidence of carcinogenicity at the dose levels tested was observed. For chronic toxicity, the

NOAEL was 1 mg/kg/day (20 mg/kg-diet) and the LOAEL was 5 mg/kg/day (100 mg/kg-diet) based on decreased body weight gains, food consumption, food efficiency, and increased blood urea nitrogen.

In a two generation reproduction study (MRID 40446201), Charles River CD rats received either 0, 2, 20, or 100 mg/kg/day technical thiobencarb (96.7% a.i.) by daily oral gavage in 0.5% CMC aqueous solution. Systemic toxicity was noted at 20 mg/kg/day and above based on enlargement of centrolobular hepatocytes (both generations) and hepatocyte single cell necrosis observed in both sexes of both generations including renal atrophic tubule consisting of regenerated epithelium. There were increased liver weights (absolute and relative) and increased kidney weights (absolute and relative) in the high dose group. There were also significant changes on body weights at 100 mg/kg/day and male kidney weights were increased in the high dose group. There were no effects on reproductive parameters. For Parental/Systemic toxicity, the NOAEL was 2 mg/kg/day and the LOAEL was 20 mg/kg/day based on histopathological changes of the liver and kidney. For reproductive toxicity, the NOAEL was equal to or greater than 100 mg/kg/day and the LOAEL was greater than 100 mg/kg/day. For the purpose of this risk assessment, the endpoints used in the risk assessment are the NOAEL of 20 mg/kg/day and corresponding LOAEL of 100 mg/kg/day based on decreased body weight. A value of ≥100 mg/kg/day was also used in characterization as a decrease in body weight does not necessarily mean a reduction in survival and reproduction may occur.

Thiobencarb was rapidly absorbed after oral administration with almost all eliminated in the urine within 72-hours (MRID 42340302).

Other sublethal effects observed in rats include gait abnormalities, decreased sensory responses, and decreased motor activity in an acute neurotoxicity screening at 500 mg/kg-day (NOAEL = 100 mg/kg-day, MRID 42987001, 43148202, acceptable).

4.4.3. Toxicity to Terrestrial Invertebrates

Terrestrial invertebrate toxicity data are used to evaluate potential indirect effects to the CRLF and to adversely modify designated critical habitat. Thiobencarb is considered practically nontoxic to honey bees (*Apis mellifera*) on an acute contact exposure basis (MRID 46059804). In this study, adult bees were exposed to a TGAI via oral and contact exposure routes at nominal concentrations of 0 and 100 µg a.i./bee. At the highest concentration tested in the oral test, 23.3% of the bees died in the treatment group as opposed to 6.7 and 3.3% in the negative and solvent control groups, respectively. In the contact exposure group, 15% of bees died as compared to 6.7 and 10% in the control groups. The 48-hr LD_{50} is thus greater than 100 µg a.i./bee for both studies. No sublethal effects were observed. This study is classified as supplemental because the age of the bees at test initiation was not reported.

The acceptable ECOTOX data were examined for toxicity data using non-target species with endpoints expressed in terms similar to those for the standard test with honey bees. No values were available. Values were reported for the brown plant hopper; however, the value reported was for an aquatic test for a snail. Two studies were available for nematodes. The LOAEL reported for the Nemata and root-knot nematode were 0.75 kg a.i./ha with the endpoint examined being population effects (Das *et al.*, 1997; Das *et al.*, 1998). These values were not reviewed for possible use in the calculation of a risk quotient.

4.4.4. Toxicity to Terrestrial Plants

Plant toxicity data from both registrant-submitted studies and studies in the scientific literature were reviewed for this assessment. Registrant-submitted studies are conducted under conditions and with species defined in EPA toxicity test guidelines. Sublethal endpoints such as plant growth, dry weight, and biomass are evaluated for both monocots and dicots, and effects are evaluated at both seedling emergence and vegetative life stages. Guideline studies generally evaluate toxicity to ten crop species. These tests are conducted on herbaceous crop species only, and extrapolation of effects to other species, such as the woody shrubs and trees and wild herbaceous species, contributes uncertainty to risk conclusions.

Commercial crop species have been selectively bred, and may be more or less resistant to particular stressors than wild herbs and forbs. The direction of this uncertainty for specific plants and stressors, including thiobencarb, is largely unknown. Homogenous test plant seed lots also lack the genetic variation that occurs in natural populations, so the range of effects seen from tests is likely to be smaller than would be expected from wild populations.

Two terrestrial plant studies with thiobencarb have been submitted to the Agency: seedling emergence studies (MRID 41690902 and 44846201) and a vegetative vigor study (MRID 41690902). For the seedling emergence and vegetative vigor testing the following plant species and groups should be tested: (1) six species of at least four dicotyledonous families, one species of which is soybean (*Glycine max*), and another of which is a root crop, and (2) four species of at least two monocotyledonous families, one species of which is corn (*Zea mays*).

Results of Tier II seedling emergence toxicity testing on technical thiobencarb are given in Table 4-5.

Table 4-5. Summary of submitted terrestrial plant Tier II seedling emergence toxicity results for thiobencarb

Species	% a.i.	Parameter Affected	EC_{25} (lb a.i./A)	NOAEC (lb a.i./A)	MRID No. Author/Year
Monocot--Corn	96.6	Shoot length	>1.7	1.7	MRID 41690902 Hoberg, J.R. 1990
Monocot--Oat		Shoot length	0.086	0.055	
Monocot--Onion		Shoot length	2.0	0.94	
Monocot--Ryegrass		Mortality	0.019	0.0051[1]	
Dicot/Root Crop--Carrot		Shoot length	>3.1	2.1	
Dicot--Cabbage		Shoot length	0.082	0.071	
Dicot-Cucumber		Shoot length	>1.7	0.16	
Dicot--Lettuce		Mortality	0.27	--	
Dicot--Soybean		Shoot length	>1.7	0.94	
Dicot--Tomato		Shoot length	1.1	0.94	
Dicot - Lettuce Monocot – Ryegrass	96.5	None (measured seedling emergence, seedling survival, plant height, dry weight, and phytotoxicity)	> 0.010 lb a.i./A.	>0.010 lb a.i./A	MRID 44846201 Chetram, R.S. 1999 Acceptable DER 11/16/2002

1 This NOAEL is based on 17% mortality of plants occurring at the next higher test level, 0.011 lb a.i./A.

In the tier II seedling emergence test, mortality of test plants occurred in the tests with ryegrass and lettuce. Mortality was the most sensitive toxic endpoint for these species (plants tended to die shortly after emerging). The most sensitive species was ryegrass, a monocot, for which the EC_{25} based on mortality (*i.e*., LC_{25}) was 0.019 lb a.i./A. The most sensitive dicot was cabbage. The cabbage EC_{25} based on shoot length was estimated to be 0.082 lb a.i./A.

Results of Tier II seedling vegetative vigor toxicity testing on the technical thiobencarb are given in Table 4-6.

Table 4-6. Summary of submitted Tier II seedling vegetative vigor toxicity testing for thiobencarb

Species	% A.I.	Parameter Affected	EC_{25} (lb a.i./A)	NOAEC (lb a.i./A)	MRID No. Author/Year
Monocot- Corn	96.6	Shoot length, shoot weight, and root weight	>2.2	2.2	MRID 41690902 Hoberg, J.R. 1990 Supplemental
Monocot--Oat		Shoot weight	0.17	0.12	
Monocot--Onion		Shoot length	1.2	0.80	
Monocot--Ryegrass		Shoot length	0.073	0.020	
Dicot/ Root Crop--Carrot		Shoot length, shoot weight, and	>2.2	2.2	

Species	% A.I.	Parameter Affected	EC_{25} (lb a.i./A)	NOAEC (lb a.i./A)	MRID No. Author/Year
		root weight			
Dicot--Cabbage		Root weight	1.2	1.4	
Dicot--Cucumber		Shoot weight and root weight	--[a]	<0.12	
Dicot--Lettuce		Root weight	1.3	0.80	
Dicot--Soybean		Shoot weight	1.2	0.80	
Dicot--Tomato		Root weight	1.8	2.2	

[a] Greater than a 25% reduction was recorded at some or all exposure levels, but the EC_{25} could not be determined because no dose-response relationship was apparent.

In the Tier II vegetative vigor tests, the cucumber and soybean were the most sensitive dicots and ryegrass was the most sensitive monocot.

Based on the results of the submitted terrestrial plant toxicity tests, it appears seedling emergence is the most sensitive for both dicots and monocots.

All open literature endpoints for terrestrial plants in the accepted ECOTOX data are higher than those measured in submitted studies. Open literature toxicity data for 'target' terrestrial plant species, which include efficacy studies, are not currently considered in deriving the most sensitive endpoint for terrestrial plants. A list of these studies are available in the ECOTOX bibliography in Appendix J.

4.5. Toxicity of Degradates

No ecological toxicity data were found in the ECOTOX database for 4-chlorobenzoic acid or 4-chlorobenzaldehyde; however, ECOSAR version 1.0 predicted aquatic toxicity endpoints greater than those predicted and measured for thiobencarb, see Appendix B. Additionally, these degradates were not considered to be of toxicological concern in the human health risk assessment completed for the RED (Lewis, 1997). The toxicity of 4-chlorobenzoic acid and 4-chlorobenzaldehyde was assumed to be less than that of thiobencarb (see Appendix B). Concentrations of the other degradates were a very small percentage (<8.3%) of the amount of thiobencarb applied or were only present in a few fate studies, indicating that they would be present at lower concentrations than those estimated for thiobencarb. The estimated toxicity based on structure activity relationships (ECOSAR version 1.0) or the similarity of the structure to thiobencarb indicates that the toxicity of these compounds is similar to or less than that of thiobencarb (see Appendix B). The presence of these degradates is not expected to alter risk conclusions that are based on the fate, transport, and toxicity of the parent compound alone.

4.6. Toxicity of Chemical Mixtures

As previously discussed, the results of available toxicity data for mixtures of thiobencarb with other pesticides are presented in Appendix C. The limited data available do no allow a comparison of the toxicity results of thiobencarb alone versus thiobencarb and propanil or with other chemicals.

4.7. Incident Database Review

A review of the EIIS database for ecological incidents involving thiobencarb was completed in September 2009. Based on the EIIS database, there have been a total of seven reported ecological incidents potentially involving thiobencarb. The incidents occurred at unreported times and between 1997 and 1998. The crop damaged in six incidents was the crop that thiobencarb was registered to be used on, rice. The damage involved an 'unknown' number of impacted acres of rice production and was described as 'damage' and 'stunted growth'. The legality of use was undetermined in four of the incidents and involved a registered thiobencarb use in three incidents. No other herbicides besides thiobencarb were involved in any of the reported incidents. The certainty that thiobencarb was responsible for the plant damage and stunted growth ranged from 'possible' (three incidents), 'probable' (two incidents), and 'highly probable' (one incident). In the remaining incident, the plant damaged was not specified.

Table 4-7. Summary of Incident Reports Involving Effects on Rice in California

INCIDENT NO.	YEAR	LEGALITY	CERTAINTY	PRODUCT	OTHER CHEMICALS INVOLVED (PC CODE)	Comments
I006793-009	NR	Undetermined	Possible	Bolero 10G	N/A	Rice crop damage observed after application of thiobencarb to rice.
I006793-007	1997	Registered Use	Probable	Bolero 10G	N/A	Distortion of rice leaves observed after application of thiobencarb to rice.
I006793-008	N/R	Registered Use	Probable	Bolero 10G	N/A	Damage to rice and a reduced yield occurred after application of thiobencarb to rice.
I007467-026	1998	Registered Use	Possible	Bolero	N/A	Damage to rice plants and reduced rice yield observed after application of thiobencarb to rice.
I004940-003	1997	Undetermined	Highly Probable	Bolero 10G	N/A	Only reported that incident reported with use of thiobencarb in an agricultural area.
I007467-025	1997	Undetermined	Possible	Bolero	N/A	Rice field leaves were distorted after application of thiobencarb.
I007776-008	1997	Undetermined	Probable	Bolero 10G	N/A	Damage to rice field observed after application of thiobencarb.

Abbreviations: NR=not reported; N/A= not applicable

4.7.1. Other Aquatic Incidents

MRID 42130705 & 42130708. Aquatic field studies also suggest that application of thiobencarb to rice paddies may result in effects to aquatic organisms. Declines in fish, aquatic invertebrates, and gravid shrimp cannot conclusively be attributed to the use of thiobencarb. Nevertheless, the findings in the field were consistent with effects demonstrated in laboratory studies. They suggest that the application of thiobencarb to rice fields may result in significant environmental damage to the adjacent estuarine habitat. Possible effects include chronic effects to sensitive fish, acute and chronic effects to ecologically important aquatic invertebrates, chronic effects to grass shrimp and possibly to commercial shrimp, and indirect detrimental effects to organisms at higher trophic levels that depend on these organisms for food.

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 CRLF and DS or for modification to their designated critical habitat from the use of thiobencarb 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").

5.1. Risk Estimation

Risk is estimated by calculating the ratio of the estimated environmental concentration (EEC) (from aquatic modeling results, T-REX for terrestrial animals, and TerrPlant for terrestrial plants) (Section 3) and the appropriate toxicity endpoint (Section 4). This ratio is the risk quotient (RQ), which is then compared to pre-established acute and chronic levels of concern (LOCs) for each category evaluated (Appendix F). For acute exposures to the aquatic animals, as well as terrestrial invertebrates, the LOC is 0.05. For acute exposures to birds (and, thus, reptiles and terrestrial-phase amphibians) and mammals, the LOC is 0.1. The LOC for chronic exposures to animals, as well as acute exposures to plants is 1.0.

In cases where the baseline RQ exceeds one or more LOC (*i.e.*, "may affect"), additional factors, including the life history characteristics of the assessed species, refinement of the baseline EECs using site-specific information, and available monitoring data are considered and used to characterize the potential for thiobencarb to adversely affect the assessed species and/or their designated critical habitat. Risk quotients used to evaluate potential direct and indirect effects to the CRLF and DS and to designated critical habitat are in Sections 5.1.1 (direct effects) and 5.1.2 (indirect effects). RQs are described and interpreted in the context of an effects determination in Section 5.2 (risk description).

5.1.1. Calculation of RQs used to Assess Direct Effects to CRLF and DS

Toxicity values used to calculate RQs are discussed in Section 4, and exposure values are discussed in Section 3. RQs used to estimate acute and chronic direct effects are in Table 5-1 (DS and aquatic-phase CRLF) and Table 5-2 (terrestrial-phase CRLF).

5.1.1.a. Direct Effects to Aquatic-phase CRLF and DS

The species considered in this risk assessment include a frog and a fish species. Direct effects to the DS are evaluated using the lowest acute and chronic toxicity values across freshwater and estuarine/marine fish species. Direct effects to the aquatic phase CRLF are evaluated using the lowest freshwater acute and chronic toxicity values across fish and amphibian toxicity studies. However, fish were consistently shown to be more sensitive than aquatic-phase amphibians and the available amphibian studies are classified as 'supplemental' and may only be used qualitatively; therefore, fish acute and chronic toxicity values are used to calculate RQs for aquatic-phase amphibians. Delta smelt may inhabit both saltwater and freshwater habitat and the most sensitive freshwater or E/M fish endpoints were used to evaluate direct risk to DS. For the CRLF, exposure was evaluated in the rice paddy. For DS, acute exposure was evaluated for exposure to water released from rice paddies after a required 14-day holding period because DS are not expected to be found in rice paddies. Chronic Exposure for the DS was estimated using time-weighted average concentrations from monitoring data.

Table 5-1. Acute and Chronic RQs for Direct Effects to the Aquatic-Phase CRLF and DS

Basis of Exposure Estimates	**Peak EEC (µg/L)**	**Chronic EEC (µg/L)**	**Risk Quotients**			
			CRLF (FW fish is surrogate)	**CRLF and DS (FW fish is surrogate)**	**DS (E/M fish is surrogate)[4]**	**DS (E/M fish is surrogate)**
			Acute[1]	**Chronic[2]**	**Acute[3]**	**Chronic[4]**
Conservative Exposure Estimate in Paddy Water for CRLF Only[5]						
Tier I Rice Model and Aquatic Dissipation for CRLF	2018	968[6]	**4.59**	**46.10**		
Conservative Exposure Estimate Where Paddy Water is Released for CRLF and DS						
Tier I Rice Model with Aquatic Dissipation and Monitoring Data	350[7]	5.37[8]	**0.80**	0.26	**1.72**	0.90

LOC exceedances (acute RQ $\geq$ 0.05; chronic RQ $\geq$ 1.0) are bolded. Acute RQ = use-specific peak EEC /96-hr LC_{50}; Chronic RQ = 60 day EEC/NOAEC

1-The freshwater vertebrate endpoint used to calculate acute RQs for the CRLF is the 96-hr LC_{50} of 440 µg/L for striped bass (ECOTOX E15472).

2-The NOAEC used to calculate chronic RQs for CRLF was 21 µg/L for the striped bass for posthatch survival (E15472).

3-The E/M fish acute endpoint used to calculate RQs for the DS is the 96-hr LC_{50} of 204 µg/L for the Atlantic Silverside (E11868).

4-The NOAEC used to calculate chronic RQs for E/M fish was 6 µg/L estimated using the ACR of 37 calculated for the striped bass and the lowest acute toxicity endpoint for Atlantic silverside (MRID 00079112 and E11868).

5-RQs for DS were not estimated in paddy water because DS are not expected to be present in rice paddies.

6- Time-weighted average concentration over 14-days water is in rice paddy. Estimated based on an initial concentration determined using the Tier I Rice Model and an aquatic dissipation rate from an aquatic dissipation study on California wet seeded rice (MRID 43404005).
7- Exposure was estimated using the Tier I Rice Model and allowing for 14 days of dissipation using the aquatic dissipation rate from MRID 43404005.
8- Time-weighted average concentration from monitoring data collected in rice growing areas of California (see Section 3.1.1.a).

Risk quotients were calculated for conservative exposure estimated using the Tier I Rice Model and modified Tier I Rice Model to account for dissipation. For the CRLF, chronic EECs were estimated based on the 14-day average concentration in the rice paddy. The 14-day value was used because water is not expected to be held in the rice paddy for more than 14-days and this is a conservative estimate for a 60-day exposure estimate. Chronic exposure outside of the rice paddy was estimated based on the time-weighted average concentration from monitoring data in California rice growing areas. All acute risk quotients for the aquatic-phase CRLF and DS exceed the LOC of 0.05 (CRLF RQs 0.80 – 4.59; DS RQs 0.80 – 1.72). Chronic RQs for the CRLF exceed the LOC of 1.0 for modeled exposure. Chronic RQs for the DS did not exceed the LOC of 1.0.

5.1.1.b. Direct Effects to Terrestrial-phase CRLF

Potential direct acute effects to the terrestrial-phase CRLF are derived by considering dose- and dietary-based EECs modeled in T-REX for a small bird (20 g) consuming small invertebrates and acute oral and subacute dietary toxicity endpoints for avian species (since no terrestrial-phase amphibian toxicity data were available for thiobencarb).

The avian acute and subacute endpoints are not definitive (*i.e.* greater than values); therefore, definitive RQs cannot be calculated. There was no mortality in the limited number of studies available and the potential for direct acute or subacute effects to the CRLF are presumed low. However, comparing the highest dose tested in toxicity studies to the EECs can provide insight into the potential for direct effects to the CRLF. The dose-based endpoint, LD_{50} >1938 mg a.i./kg-bw (bobwhite quail), is higher than the dose based EECs of 615 mg/kg-bw (MRID 42600201 and Table 3-6). The ratio of the EECs to the highest dose tested is 0.44. The subacute dietary endpoint, LC_{50} >5000 mg a.i./kg-bw is well above the highest dietary based EEC of 540 mg/kg-diet (MRID 44846206 and Table 3-6). The ratio of the EEC to the highest dietary concentration tested is 0.11. If toxicity were observed slightly above the highest levels tested, there is a potential that the LOC for listed species (0.1) could be exceeded. The EEC estimated for granular formulations was 42 mg a.i./ft^2. The adjusted avian LD_{50}/ft^2, comparable to an RQ, is <1.49 (Appendix G). This value is greater than the acute avian LOC of 0.1.

Potential direct chronic effects of thiobencarb to the terrestrial-phase CRLF are derived by considering dietary-based exposures modeled in T-REX for a small bird (20g) consuming small invertebrates. Chronic effects are estimated using the lowest available toxicity data for birds. Chronic reproductive effects to birds were observed at 300 mg a.i./kg-diet (LOAEC), with an associated NOAEC of 100 mg a.i./kg-diet (MRID 00025778). The T-REX estimated chronic RQ of 9.60 exceeds the chronic LOC of 1.0. These RQs are further characterized in the context of the effects determination in Section 5.2.

5.1.2. Calculation of RQs used to Assess Indirect Effects to CRLF and DS

This section presents RQs used to evaluate the potential for thiobencarb to induce indirect effects. Pesticides have the potential to exert indirect effects upon listed species by inducing changes in structural or functional characteristics of affected communities. Perturbation of forage or prey availability and alteration of the extent and nature of habitat are examples of indirect effects. A number of these indirect effects are also considered as part of the critical habitat modification evaluation. In conducting a screen for indirect effects, direct effects LOCs for each taxonomic group (*e.g.*, freshwater fish, invertebrates, aquatic plants, and terrestrial plants) are employed to make inferences concerning the potential for indirect effects upon listed species that rely upon non-listed organisms in these taxonomic groups as resources critical to its life cycle (USEPA, 2004). This approach used to evaluate indirect effects to listed species is endorsed by the Services (USFWS/NMFS/NOAA, 2004). If no endangered species LOCs are exceeded for organisms on which the assessed species depends for survival or reproduction, indirect effects are not expected to occur.

If LOCs are exceeded for organisms on which the assessed species depends for survival or reproduction, dose-response analysis is used to estimate the potential magnitude of effect associated with an exposure equivalent to the EEC. The greater the probability that exposures will produce effects on a taxa, the greater the concern for potential indirect effects for listed species dependant upon that taxa (USEPA, 2004).

As an herbicide, indirect effects to the assessed species from potential effects on primary productivity of aquatic plants are a principle concern. If plant RQs fall between the risk to endangered species and non-endangered species LOCs, a no effect determination is made for listed species that rely on multiple plant species to successfully complete their life cycle (termed plant dependent species). If plant RQs are above risk to non-endangered species LOCs, this could be indicative of a potential for adverse effects to those listed species that rely either on a specific plant species (plant species obligate) or multiple plant species (plant dependant) for some important aspect of their life cycle (USEPA, 2004). Based on the information provided in Section 2.6, the assessed species do not have any known obligate relationship with a specific species of aquatic plant.

Direct effects to riparian zone vegetation may also indirectly affect the assessed species by reducing water quality and available spawning habitat via increased sedimentation. Direct impacts to the terrestrial plant community (*i.e.*, riparian habitat) are evaluated using submitted terrestrial plant toxicity data. If terrestrial plant RQs exceed the Agency's LOC for direct risk to non-endangered plant species, based on EECs derived using EFED's Terrplant model (Version 1.2.1), a conclusion that thiobencarb may affect the CRLF and DS via potential indirect effects to the riparian habitat (and resulting impacts to habitat due to increased sedimentation) is made. Further analysis of the potential for thiobencarb to affect the CRLF and the DS via reduction in riparian habitat includes a description of the importance of riparian vegetation to the assessed species and types of riparian vegetation that may potentially be impacted by thiobencarb use within the action area.

RQs used to evaluate the potential for thiobencarb to induce indirect effects to the assessed species are presented in Sections 5.1.2.a to 5.1.2.e. These RQs suggest that potential indirect effects could occur by potentially impacting food availability and primary productivity as indicated by LOC exceedances. These RQs were based on the most sensitive surrogate species tested across aquatic invertebrate, fish, and aquatic plant species tested. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.a. Aquatic Vertebrates

Indirect effects to the adult CRLF may result from reduction in aquatic vertebrate prey items. Risk quotients for freshwater vertebrates were previously estimated in Section 5.1.1.a. Since the acute and chronic RQs are exceeded, there is a potential for indirect effects to listed species that rely on freshwater fish and/or aquatic-phase amphibians (*e.g.*, CRLF) during at least some portion of their life-cycle.

5.1.2.b. Aquatic Invertebrates

Risk quotients were calculated exposure estimated using the Tier I Rice Model and modified Tier I Rice Model to account for dissipation. For the CRLF, chronic EECs were estimated based on the 14-day average concentration in the rice paddy. The 14-day value was used because water is not expected to be held in the rice paddy for more than 14-days and this is a conservative estimate for a 60-day exposure estimate. Chronic exposure outside of the rice paddy was estimated based on the time-weighted average concentration from monitoring data in California rice growing areas. Acute risk to aquatic E/M invertebrates is estimated in the same way as those estimated for freshwater invertebrates except that risk quotients were not calculated for exposure in rice paddies as E/M invertebrates are not expected to be present in rice paddies. The EECs are compared to the lowest acute toxicity value for freshwater or E/M invertebrates. Chronic risk for the E/M invertebrates is based on the time-weighted average concentration estimated using monitoring data.

Table 5-2. Acute and Chronic RQs for Aquatic Invertebrates Used to Evaluate Potential Indirect Effects to the CRLF and the DS Resulting from Potential Impacts to Food Supply

Basis of Exposure Estimates	Peak EEC (µg/L)	Chronic EEC (µg/L)	Risk Quotients			
			FW Invertebrates		E/M Invertebrates[5]	
			Acute[1]	Chronic[2]	Acute[3]	Chronic[4]
Conservative Exposure Estimate in Paddy Water (CRLF Only)						
Tier I Rice Model and Aquatic Dissipation	2018	968[5]	**19.94**	**968**	--	--
Conservative Exposure Estimate Where Paddy Water is Released (CRLF and DS)						
Tier I Rice Model with Aquatic Dissipation and Monitoring Data	350[6]	5.37[7]	**3.46**	**5.37**	**2.33**	**1.68**

LOC exceedances (acute RQ $\geq$ 0.05; chronic RQ $\geq$ 1.0) are bolded. Acute RQ = peak EEC /96-hr LC_{50}; Chronic RQ = 21 day EEC/NOAEC

1- The freshwater invertebrate endpoint used to calculate acute RQs is the 96-hr LC_{50} of 101.2 µg/L for *Daphnia magna* (MRID 00025788).
2- The NOEAC used to calculate chronic RQs for freshwater invertebrates is 1 µg/L for the *Daphnia magna* based on reduced number of offspring produced (MRID 00079098).
3- The E/M acute endpoint used to calculate RQs is the 96-hr LC_{50} of 150 µg/L for the Mysid shrimp (MRID 00050667).
4- The E/M acute endpoint used to calculate RQs is the 21-day NOAEC of 3.2 µg/L for the Opossum shrimp based on reduced survival of offspring (MRID 43976801 or 40651314).
5- 14-day average concentration in rice paddy estimated based on an initial concentration determined using the Tier I Rice Model and an aquatic dissipation rate from an aquatic dissipation study on California wet seeded rice (MRID 43404005).
6- Exposure was estimated using the Tier I Rice Model and allowing for 14 days of dissipation using the aquatic dissipation rate from MRID 43404005.
7- The chronic values represents the average of concentrations measured between May 9 and June 22, 2000 in the NAWQA monitoring data collected in rice growing areas of California.

Based on exceedances of the Agency's acute listed species LOC (RQ>0.05) and chronic risk LOC (RQ>1) using modeled EECs and using thiobencarb concentrations measured in the Sacramento River, there is a potential for indirect effects on the CRLF and indirect effects to the DS, as they rely on freshwater invertebrates and/or E/M invertebrates during at least some portion of their life-cycle. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.c. Terrestrial Invertebrates

In order to assess the risks of thiobencarb to terrestrial invertebrates, which are considered prey of CRLF in terrestrial habitats, the honey bee is used as a surrogate for terrestrial invertebrates. The toxicity value for terrestrial invertebrates is calculated by multiplying the lowest available acute contact LD_{50} greater than 100 µg a.i./bee by 1 bee/0.128g, which is based on the weight of an adult honey bee (MRID 46059804). EECs (µg a.i./g of bee) calculated by T-REX for small and large insects are divided by the calculated toxicity value for terrestrial invertebrates, which is greater than 781 µg a.i./g of bee. Terrestrial invertebrate EEC/highest dose tested are presented in Table 5-3.

Table 5-3. Summary of EEC/Highest Dose Tested Ratio for Terrestrial Invertebrates on the Site of Application - Used to Evaluate Potential Indirect Effects to the CRLF Resulting from Potential Impacts to the Food Supply.

Use	Application Rate (lb a.i./acre)	Size Class	EEC (ppm)	EEC/Highest Dose Tested Ratio[1]
Rice	4	Small insect	540	**0.69**
		Large insect	60	**0.08**

1 Available acute contact toxicity data for bees exposed to thiobencarb (in units of µg a.i./bee) are converted to µg a.i./g (of bee) by multiplying by 1 bee/0.128 g (LD_{50} >100 µg a.i./bee becomes >781 µg/g bee).
2 Bolded ratios potentially exceed the interim risk to listed species LOC for terrestrial invertebrates (0.05).

In the honey bee contact study, 15% of bees died at the highest dose tested (100 µg a.i./bee) as compared to 6.7 and 10% in the control groups (MRID 46059804). A definitive LD_{50} value was not determined. The RQs for all thiobencarb uses are potentially above the Agency's interim risk to listed species LOC for terrestrial invertebrates (0.05). The actual RQs would likely be lower than the ratio of the EEC to the highest dose tested reported in Table 5-3. However, it is not clear if the actual RQs would be above or below the Agency's LOC without definitive data, therefore, risks cannot be precluded at this time. Discussion of these RQs in the context of this effects determination is presented in Section 5.2.

5.1.2.d. Mammals

Potential risks to mammals are derived using T-REX and acute and chronic mammal toxicity data. RQs are typically derived for various sizes of mammals (15 g, 35 g, and 1000 g); however, RQs are not presented for 1000 g mammals because it is improbable that even the largest CRLF would consume a mammal of that size. Therefore, the evaluation for potential indirect effects to the CRLF resulting from potential reductions in mammal abundance as food is based on the 15 g size class, which results in higher RQs than the 35 g mammal. The California mouse (*Peromyscus californicus*) is a particular species known to be consumed by the CRLF. The California mouse is omnivorous and consumes grasses, fruits, flowers, and invertebrates (University of South Carolina, 2005). Therefore, the short grass food item was used to determine if mammals could be impacted; however, RQs based on EECs on other food items were also derived for characterization purposes. A range of RQs for mammals is presented in Table 5-4 (also see Appendix G).

Table 5-4. Summary of Acute and Chronic RQs for 15 g Mammals Used to Evaluate Potential Indirect Effects to the CRLF Resulting from Potential Impacts to the Food Supply.[1]

Dietary Category	Dose Based EEC (mg/kg-bw)	Dietary Based EEC (mg/kg-diet)	Acute Dose-Based RQ[2]	Chronic Dose Based RQ[3]	Chronic dietary-Based RQ[3]
On Field Liquid Broadcast Applications (Aerial or Ground)[1]					
Short grass	915.29	960.00	**0.40**	**416.45**	**48.00**
Tall grass	419.51	440.00	**0.18**	**190.87**	**22.00**
Broadleaf plants/small insects	514.85	540.00	**0.23**	**234.25**	**27.00**
Fruits/pods/large insects	57.21	60..00	0.03	**26.03**	**3.00**
Seeds	12.71		0.01	**5.78**	
Granular Applications[4]					
n/a	42 mg a.i./ft^2	**n/a**	**1.22**	**n/a**	**n/a**

Abbreviations: n/a=not applicable
Bolded RQs exceed (or are near) the acute listed species LOC (0.1) or chronic listed species LOC (1) for mammals.
1- Estimated residues for potential food items using T-REX.
2- The acute oral LD_{50} of 1033 mg a.i./kg-bw for the rat was used to calculate acute dose based RQs (MRID 42130701).
3- The NOAEL of 1 mg/kg-bw for the Fischer rat was used to calculate chronic RQs (MRID 40446201). The effects observed were decreased body weight gain, food consumption, and food efficiency.

RQs exceed acute LOCs for mammals consuming short grass, tall grass, and broadleaf plants or small insects. Chronic RQs are exceeded for all diet classes. Based on an LD50/ft^2 analysis, RQs for use of granular formulation also exceed the acute LOC of 0.1. A method to evaluate chronic risk to granular formulations is currently not available. Therefore, the RQs estimated for liquid formulations are assumed to represent chronic exposure to both liquid and granular formulations.

5.1.2.e. Aquatic and Terrestrial Plants

Aquatic plants serve as food supply for both the CRLF and the DS and can impact water quality. Additionally, effects to terrestrial plants can impact terrestrial habitat quality and water quality parameters. Therefore, RQs for vascular and non-vascular aquatic plants are used to evaluate the potential for thiobencarb to affect the CRLF and/or the DS by potentially impacting the food supply and water quality, and thus habitat. RQs for terrestrial plants are used to evaluate the potential for thiobencarb to impact aquatic habitats (*i.e.*, water quality) (aquatic-phase CRLF and DS) and/or terrestrial habitats (terrestrial-phase CRLF).

i. Aquatic Plants

Risk to aquatic non-vascular plants is based on peak EECs and the lowest 120-hr EC_{50} value for non-vascular plants (17 µg/L for the Green algae; MRID 41690901) or the lowest 14-day EC_{50} for vascular plants (770 µg/L for Duckweed; MRID 41690901). Table 5-5 shows estimated RQs for aquatic plants.

Table 5-5. Summary of RQs for Vascular and Non-Vascular Aquatic Plants.

Basis of Exposure Estimates	Peak EEC (µg/L)	Risk Quotients	
		Non-vascular Plants[1]	Vascular Plants[2]
Conservative Exposure Estimate in Paddy Water (CRLF)			
Tier I Rice Model and Aquatic Dissipation	2018	**118.71**	**2.62**
Conservative Exposure Estimate Where Paddy Water is Released (CRLF and DS)			
Tier I Rice Model with Aquatic Dissipation and Monitoring Data	350[3]	**20.59**	0.45

LOC exceedances (RQ $\geq$ 1.0) are bolded. RQ = peak EEC /5-day EC_{50} for nonvascular plants or 14-day EC_{50} for vascular plants.

1- The non-vascular plant endpoint was used to calculate RQs for the 5-day EC_{50} of 17 µg/L for the Green algae (MRID 41690901).
2- The vascular plant endpoint used to calculate acute RQs is the 14-day EC_{50} of 770 µg/L for Duckweed (MRID 41690901).
3- Exposure was estimated using the Tier I Rice Model and allowing for 14 days of dissipation using the aquatic dissipation rate from MRID 43404005.

LOCs for vascular plants were only exceeded based on exposure estimated in the rice paddy. LOCs for non-vascular plants were exceeded using EECs estimated for the rice paddy but not for exposure estimates allowing for fourteen days dissipation. Since the RQs are exceeded, there is a potential for indirect effects to the CRLF and DS that rely on non-vascular aquatic plants during at least some portion of their life-cycle. Indirect effects to the CRLFmay occur due to effects to vascular plants in rice paddies.

ii. Terrestrial Plants

Potential indirect effects resulting from effects on terrestrial vegetation were assessed using RQs from terrestrial plant seedling emergence and vegetative vigor EC_{25} data as a screen. Based on the results of the submitted terrestrial plant toxicity tests, emerging seedlings are more sensitive to thiobencarb via soil/root uptake than emerged plants via foliar routes of exposure. Therefore, the seedling emergence data were used to estimate terrestrial plant RQs for thiobencarb use. RQs used to estimate potential indirect effects to the CRLF and/or the DS from potential effects to terrestrial plants within their habitat areas are summarized in Table 5-6.

Table 5-6. Non-Listed Terrestrial Plant RQs for Thiobencarb Use on Rice (one application at 4 lbs a.i./acre).[1]

Type of Application, Formulation	Drift Only	
	Monocot	Dicot
Aerial, Liquid	**10.53**	**2.44**
Ground, Liquid	**2.11**	0.49
Aerial and Ground, Granular	<0.1	<0.1

RQs greater than the LOC of 1.0 are shown in bold.
[1] Based on the following: monocots - EC_{25} = 0.019 lb a.i./acre (seedling emergence) and EC_{25} = 0.073 lb a.i./acre (vegetative vigor) in ryegrass; dicots - EC_{25} = 0.082 lbs a.i./acre [seedling emergence, cabbage)] and EC_{25} = 1.2 lbs a.i./acre [vegetative vigor, cabbage] (MRID 41690902).

Based on available data, monocots are more sensitive to thiobencarb than are dicots. RQs of monocots exceed the LOC for non-listed plants (1.0) for liquid applications. Granular applications are not exceeded because spray drift from application of granular formulations is expected to be minimal and runoff is not expected to be a concern in rice paddies. For Dicots, RQs are exceeded for aerial applications of a liquid formulation but not for ground applications. LOCs were exceeded for both dicot and monocot terrestrial plants, which could result in indirect effects to the CRLF or the DS. These LOCs and their impact on the effects determination are described in Section 5.2.

5.1.3. Primary Constituent Elements of Designated Critical Habitat

For thiobencarb use, 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 presented in Section 5.1 are used as the basis of the effects determination for potential modification to designated critical habitat.

5.2. Risk Description

The risk description synthesizes overall conclusions regarding the likelihood of adverse impacts leading to an 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.

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 "no effect" determination is made, based on thiobencarb'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 thiobencarb. Risk quotients indicate that use of thiobencarb on rice paddies may affect both the CRLF and DS with both direct and indirect effects. Additionally, modification of PCEs and critical habitat may occur. A summary of the risk estimation results are provided in Table 5-7 for direct and indirect effects to the listed species and for the PCEs of their designated critical habitat.

Table 5-7. Summary of Risk Estimation for Thiobencarb. The Risk Estimation Is Further Refined in the Risk Description

Listed Species Taxonomic Group of Concern	Effects Associated with RQ	Risk Quotients (RQs) or Ratios	Description	Assessed Species Potentially Affected	
				Direct Effects	Indirect Effects
Aquatic Environment					
Freshwater fish in Paddy (surrogate for CRLF)	Acute: mortality	4.59	All RQs exceed LOC of 0.05.	CRLF	CRLF
	Chronic: Posthatch survival	46.10	RQs exceed LOC of 1.0 for 14-day average modified Tier I Rice model estimate.	CRLF	CRLF
Freshwater fish in Tailwater (surrogate for CRLF and DS)	Acute: mortality	0.80	All RQs exceed LOC of 0.05.	DS	--
	Chronic: Posthatch survival	0.26	RQ does not exceed the LOC of 1.0 based on monitoring data	--	--
Estuarine/Marine fish	Acute: mortality	1.72	All RQs exceed LOC of 0.05.	DS	--
	Chronic: wet weight	0.90	RQ does not exceed the LOC of 1.0 based on monitoring data	--	--
Freshwater invertebrates in paddy water and where paddy water is released	Acute: mortality	3.46 - 19.94	All RQs exceed the LOC of 0.05.	--	CRLF and DS
	Chronic: Offspring produced	5.37 - 968	All RQs exceed the LOC of 1.0.	--	CRLF and DS
Estuarine/Marine Invertebrates	Acute: mortality	2.33	RQ exceeds the LOC of 0.05.	--	DS
	Chronic: Posthatch survival	1.68	RQ exceeds the LOC of 1.0.	--	DS
Non-listed Aquatic vascular plants	Frond production	0.45 – 2.62	Only RQs in the rice paddy exceed the LOC of 1.0.	--	CRLF
Non-listed Aquatic non-vascular plants	Cell Density/ Inhibition of Growth	20.59 – 118.71	All RQs exceed the LOC of 1.0	--	CRLF and DS

Listed Species Taxonomic Group of Concern	Effects Associated with RQ	Risk Quotients (RQs) or Ratios	Description	Assessed Species Potentially Affected	
				Direct Effects	**Indirect Effects**
Terrestrial Environment					
CRLF (based on the bird as a surrogate)	Acute: No mortality	<0.11 - <1.49	Uncertain, no definitive endpoints available.	Uncertain	Uncertain
	Chronic: Reproduction	9.6	RQ exceeds LOC of 1.0 for liquid formulations. A method to evaluate chronic risk of granular formulations is not available.	CRLF	CRLF
Mammals	Acute: mortality	0.01 - 0.40 (liquid) 1.22 (granular)	RQs for mammals consuming short grass, tall grass, broadleaf plants, and small insects exceed LOC of 0.1 for both liquid and granular formulations.	--	CRLF
	Chronic: Reproductive	3.00 416.45	All RQs exceed the chronic LOC of 1.0 for liquid formulations. A method to evaluate chronic risk of granular formulations is not available.	--	CRLF
Terrestrial invertebrates	No mortality	<0.08 - <0.69	Uncertain, no definitive endpoints available.	--	Uncertain
Non-listed Dicot terrestrial plants	Seedling Emergence Mortality	<0.1 - 2.44	RQs for aerial application of liquid formulations exceed the LOC of 1.0	--	CRLF and DS
Non-listed Monocot terrestrial plants	Seedling Emergence Shoot Length	<0.1 - 52.63	RQs for aerial and ground application of liquid formulations exceed the LOC of 1.0	--	CRLF and DS

Following a "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 through 5.4.2.

5.2.1. Direct Effects

5.2.1.a. DS and Aquatic-Phase CRLFs

Acute and chronic risk quotients for both the aquatic-phase CRLF and DS exceed the LOCs for modeled EECs in rice paddies (Table 5-1). All acute RQs estimated based on modeled concentrations where paddy water is released also exceed the listed species LOC of 0.05. Acute RQs for CRLF range from 0.80 - 4.59 and acute RQs for DS ranged from 0.80 – 1.72 (based on freshwater and E/M species).

The chance of individual effect (mortality) to the aquatic-phase CRLF can be determined as described above in Section 2.10.2.b using the acute LOC (0.05) and the RQs calculated from water column EECs as threshold values for effects. For the CRLF, at the highest RQ (4.59) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.0 (estimated using the Individual Effect Chance Model (IEC) Version 1.1). For the DS, at the highest RQ (1.72) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.17 (estimated using IEC Version 1.1). Probit dose-response analysis for direct effects to CRLF and DS are shown in Table 5-8. Some acute data are available for amphibians that suggest that amphibians are less sensitive than fish. The estimated 96-hr LC_{50} values were near 1.3-6.5 mg/L. These data are highly uncertain and cannot be used to discount risk. No incidents were reported for fish or amphibians.

Table 5-8. Probit Dose-Response Analysis for Direct Effect to the CRLF and DS

Species Represented	Maximum RQ	Acute Effect Slope (95% C.I.)	Chance of Individual Effect at Listed Species LOC (95% C.I.)	Chance of Individual Effect at Derived Acute RQ[1] (95% C.I.)
CRLF (Surrogate Freshwater fish)	4.59	Mortality Default Slope = 4.5 (2 – 9)	1 in 4.18E+08 (1 in 216 to 1 in 1.75E+31)	1 in 1.0 (1 in 1.10 to 1 in 1.0)
DS (Surrogate FW or E/M Fish)	1.72	Mortality Default Slope = 4.5 (2 – 9)	1 in 4.18E+08 (1 in 216 to 1 in 1.75E+31)	1 in 1.17 (1 in 1.47 to 1 in 1.02)

Risk quotients were also estimated based on data from the aquatic dissipation study resulting in the highest exposure estimate in California and based on monitoring data to further characterize the potential for risk to the CRLF and DS from use of thiobencarb on rice. These calculations are shown in Table 5-9. All acute RQs for the CRLF and DS still exceed the acute listed species LOC of 0.05 when based on monitoring data. RQs ranged from 0.39 – 2.82.

Table 5-9. Acute and Chronic RQs for Direct Effects to the Aquatic-Phase CRLF and DS based on Aquatic Dissipation Study Results and Monitoring Data

Basis of Exposure Estimates	Peak EEC (µg/L)	Chronic EEC (µg/L)	Risk Quotients			
			CRLF (FW fish is surrogate)	CRLF and DS (FW fish is surrogate)	DS (E/M fish is surrogate)[4]	E/M fish
			Acute[1]	Chronic[2]	Acute[3]	Chronic[4]
Realistic Exposure Estimate in Paddy Water and Where Paddy Water is Released (CRLF and DS)						
Aquatic Dissipation Study (MRID 43404005)	438	80[5]	**1.00**	**3.81**	**2.15**	**13.33**
Aquatic Dissipation Study (Ross and Sava, 1986)	576	70[5]	**1.31**	**3.33**	**2.82**	**11.67**
Exposure Estimates Based on the Measured Concentration in Rice Growing Areas of CA (CRLF and DS)						
Colusa Basin Drain on Sacramento River	170[6]	5.37[7]	**0.39**	0.26	**0.83**	0.90

LOC exceedances (acute RQ $\geq$ 0.05; chronic RQ $\geq$ 1.0) are bolded. Acute RQ = use-specific peak EEC /96-hr LC_{50}; Chronic RQ = 60 day EEC/NOAEC

1- The freshwater vertebrate endpoint used to calculate acute RQs for the CRLF is the 96-hr LC_{50} of 440 µg/L for striped bass (ECOTOX E15472).
2- The NOAEC used to calculate chronic RQs for CRLF was 21 µg/L for the striped bass for posthatch survival (ECOTOX number E15472).
3- The E/M fish acute endpoint used to calculate RQs for the DS is the 96-hr LC_{50} of 204 µg/L for the Atlantic Silverside (ECOTOX E11868).
4- The NOAEC used to calculate chronic RQs for E/M fish was 6 µg/L estimated using the ACR of 34 calculated for the Sheepshead minnow and the lowest acute toxicity endpoint for Atlantic silverside (MRID 00079112 and E11868).
5- Aquatic dissipation chronic values are average 60-day concentrations measured in rice paddies. Values are only slightly lower when factoring in the 14 day holding period and a separate estimate was not prepared for this characterization.
6- The peak value represents the highest measured concentration in rice growing areas of California.
7- The chronic value represents the average of concentrations measured between May 9 and June 22, 2000 in the NAWQA monitoring.

Chronic RQs for freshwater fish were calculated based on a NOAEC of 21 µg a.i./L for the striped bass (E15472). At 23µg/L posthatch survival was reduced. The chronic endpoint may underestimate risk because hatching success was not evaluated in the study. The toxicity endpoint for E/M fish was estimated using an ACR. Chronic RQs for the CRLF and DS exceed the LOC of 1.0 when calculated based on concentrations measured in rice paddies and estimated using the Tier I Rice Model. The chronic RQ for the CRLF based on the Tier I Rice Model was 46.10. Chronic RQs for the CRLF and DS, respectively, based on concentrations measured in rice paddies were 3.33 – 3.81 and 11.67 – 13.33. These concentrations may be expected to occur in the rice paddy and near where water is released from the rice paddy. Chronic RQs for the DS and CRLF estimated based on monitoring studies indicate that chronic risk to DS and CRLF is less likely as the released paddy water is diluted and thiobencarb will have time to dissipate from the water column and the RQs were below LOCs.

These results indicated that direct effects to the CRLF and DS may occur with use of thiobencarb on rice. Risks are predicted even when using measured concentrations in rice growing areas (see Section 3.1.1 for a description of monitoring data). The likelihood of risk is further supported by aquatic field studies which suggest that application of thiobencarb to rice fields may result in significant environmental damage to fish (MRID 42130705 and 42130708).

Based on the CADPR PUR data, from 1999 to 2006 an average of 308, 491 – 1,006, 327 lbs of thiobencarb per year were applied to rice in California. Usage of thiobencarb on rice in California is expected to occur and thiobencarb is often used at the maximum application rate (Table 2-6). Finally, monitoring data are shown to be highest in May and June (Orlando and Kuivila, 2004). In May and June, the life-stage of CRLF would be young juveniles and DS may be spawning and have moved into freshwater (Table 2-7). Finally, CRLF critical habitat and DS critical habitat overlap with the cultivated crop NLCD land cover class.

Therefore, the weight of evidence based on the currently available data suggests that direct effects to aquatic-phase CRLFs and DS are expected from the use of thiobencarb on rice in California.

5.2.1.b. Terrestrial-Phase CRLF

Acute RQs for birds could not be calculated as no mortality occurred at the highest doses tested. The ratio of the EECs to the highest dose tested is 0.44. The subacute dietary endpoint, LC_{50} >5000 mg a.i./kg-bw is well above the highest dietary based EEC of 540 mg/kg-diet (MRID 44846206 and Table 3-6). The ratio of the EEC to the highest dietary concentration tested is 0.11. If toxicity were observed slightly above the highest levels tested, there is a potential that the LOC for listed species (0.1) could be exceeded. The EEC estimated for granular formulations was 42 mg a.i./ft^2. The adjusted avian LD_{50}/ft^2, comparable to an RQ, is <1.49 (Appendix G). This value could be greater than the acute avian LOC of 0.1.

Chronic reproductive effects to birds were observed at 300 mg a.i./kg-diet (LOAEC), with an associated NOAEC of 100 mg a.i./kg-diet (MRID 00025778). Effects observed included effects on eggs laid and hatchlings per live three week embryo. Sublethal effects observed in the study at concentrations of 30 mg/kg-diet include loss of coordination, lower limb weakness, prostate posture, loss of reflexes, and wing droop. The T-REX estimated chronic RQ of 9.60 exceeds the chronic LOC of 1.0. This also indicates that directs effects to the CRLF are likely to occur with use of liquid formulations of thiobencarb on rice. A method is not available to evaluate chronic risk to birds from the use of granular formulations.

Birds were used as surrogate species for terrestrial-phase CRLFs. Terrestrial-phase amphibians are poikilotherms, which means that their body temperature varies with environmental temperature, while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). As a consequence, the caloric requirements of terrestrial-phase amphibians are markedly lower than 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 relatively comparable slopes to the allometric functions, one can see that, given a comparable body weight, the free-living metabolic rate (FMR) of birds can be 40 times higher than reptiles, though the requirement differences narrow with high body weights.

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. For this not to be the case, terrestrial-phase amphibians would have to be 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 thiobencarb are lacking. To quantify the potential differences in food intake between birds and terrestrial-phase CRLFs, 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 CRLF were evaluated. These functions were encompassed in a model called T-HERPS. T-HERPS is available at: http://www.epa.gov/oppefed1/models/terrestrial/index.htm.

For the uses with the highest application rates (4 lb a.i./acre), none of the acute dose-based RQs for terrestrial herpetofauna exceed the Agency's listed species LOC for acute exposure (Table 5-10). Based on these calculations use of thiobencarb on rice is not expected to result in acute direct effects to terrestrial-phase CRLF (see also Appendix G).

Table 5-10. Upper Bound Kenaga, Acute Terrestrial Herpetofauna Dose-Based Risk Quotients for Thiobencarb (4 lb a.i./acre, 1 Application)

Size Class (grams)	Highest Dose Tested (mg a.i./kg-bw)	EECs (mg/kg-bw) and EEC/Highest Dose Tested Ratio									
		Broadleaf Plants/ Small Insects		Fruits/Pods/ Seeds/ Large Insects		Small Herbivore Mammals		Small Insectivore Mammal		Small Amphibians	
		EEC	Ratio	EEC	Ratio	EEC	Ratio	EEC	Ratio	EEC	Ratio
1.4	1938.00	20.98	<0.01	2.33	0.00	N/A[1]	N/A	N/A	N/A	N/A	N/A
37	1938.00	20.62	<0.01	2.29	0.00	N/A	N/A	N/A	N/A	0.72	0.00
238	1938.00	13.51	<0.01	1.50	0.00	57.69	<0.03	3.61	0.00	0.47	0.00

[1] N/A = not applicable (a 1.4 or 37 g frog is not expected to be large enough to eat a 35 g mammal).

At the 4 lb a.i./acre application rate the sub-acute EEC/highest concentration tested ratios for terrestrial herpetofauna that eat broadleaf plants/small insects and small herbivorous mammals have the potential to exceed the Agency's listed species LOCs based on dietary exposure to small insects and small herbivore mammals (Table 5-11). Because a definitive LC_{50} was not determined for birds (*i.e.*, the LC_{50} >5000 mg a.i./kg-diet) all of the calculated EEC/Highest Concentration Tested Ratios for sub-acute dietary exposure are less than values. The available data indicate that acute risk to birds is unlikely. However, the possibility that LOCs could be exceeded exists. Additionally, only limited data are available on the acute toxicity of thiobencarb to birds. No data are available for passerine species. Therefore, risks to terrestrial-phase frogs from acute exposure cannot be precluded for any of the uses or dietary categories at this time.

For chronic exposure, use of thiobencarb on rice results in RQs that exceed the Agency's chronic listed species LOC of 1.0 for amphibians consuming broadleaf plants/small insects and small herbivore mammals using an avian NOAEC of 100 mg a.i./kg-diet (Table 5-11). RQs range from 0.19 – 9.15. Therefore, chronic risk to the CRLF may occur with the use of thiobencarb on rice.

Table 5-11. Upper Bound Kenaga, Sub-Acute and Chronic Terrestrial Herpetofauna Dietary-Based Risk Quotients for Thiobencarb (1 Application at 4 lbs a.i./acre).

Highest Dose Tested or Endpoint (mg a.i./kg-diet)	**EECs (mg/kg-diet) and EEC/ Highest Concentration Tested Ratio or RQ**									
	Broadleaf Plants/ Small Insects		**Fruits/Pods/ Seeds/ Large Insects**		**Small Herbivore Mammals**		**Small Insectivore Mammals**		**Small Amphibians**	
	EEC	**Ratio**	**EEC**	**Ratio**	**EEC**	**Ratio**	**EEC**	**Ratio**	**EEC**	**Ratio**
Sub-Acute (Dietary)										
LC_{50} > 5000	540.00	**<0.11**	60.00	<0.01	915.29	**<0.18**	57.21	<0.01	18.74	<0.00
	Chronic (Dietary)									
	EEC	**RQ**	**EEC**	**RQ**	**EEC**	**RQ**	**EEC**	**RQ**	**EEC**	**RQ**
NOAEC = 100	540.00	**5.40**	60.00	0.60	915.29	**9.15**	57.21	0.57	18.74	0.19

- Bolded numbers (black) exceed the Agency's listed species acute LOC of 0.1 or chronic LOC of 1.0.
- Bolded numbers with a less than sign potentially exceed the Agency's acute LOC of 0.1.

These results indicate that the risk of direct adverse effects to terrestrial-phase CRLF from acute oral or sub-acute dietary exposure is likely low. However, risk cannot be precluded because limited data are available on the acute toxicity of thiobencarb to birds and the estimated ratios have the potential to exceed LOC values. Chronic risk to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for the dietary classes of small insects and small mammals.

5.2.2. Indirect Effects, DS and Aquatic-Phase CRLF

As discussed in Section 2, the diet of aquatic-phase CRLF tadpoles and DS larvae is composed primarily of unicellular aquatic plants (*i.e.*, algae and diatoms) and detritus. However, aquatic invertebrates are also consumed by both CRLFs and the DS, and fish are consumed by adult CRLFs. Therefore, potential impacts to each of these potential food items are evaluated.

5.2.2.a. Potential Impacts to Fish (Indirect Effects to CRLF Only)

Fish are food items of the CRLF. As discussed in Section 5.2.1.a, all RQs exceed LOCs for freshwater fish. Therefore, indirect effects to CRLF from a decline in potential fish prey are expected from the use of thiobencarb on rice in California.

5.2.2.b. Potential Impacts to Aquatic Invertebrates

CRLF and DS (Freshwater Aquatic Invertebrates)

Aquatic invertebrates are a potential food item for the CRLF and DS. The acute risk to listed and non-listed species LOCs of 0.05 and 0.5 were exceeded for freshwater invertebrates for thiobencarb use based on toxicity values from the most sensitive freshwater invertebrates for which data are available. The highest acute RQ is 19.94 and was estimated in rice paddies based on an endpoint for *Daphnia*. At this RQ and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.0 (estimated using IEC version 1.1, see Table 5-12).

Table 5-12. Probit Dose-Response Analysis for Aquatic Invertebrates

Species Represented	Maximum RQ	Acute Effect Slope (95% C.I.)	Chance of Individual Effect at Listed Species LOC (95% C.I.)	Chance of Individual Effect at Derived Acute RQ[1] (95% C.I.)
Freshwater Invertebrate (Indirect effects to CRLF)	19.94	Mortality Default Slope = 4.5 (2 – 9)	1 in 4.18E+08 (1 in 216 to 1 in 1.75E+31)	1 in 1.0 (1 in 1.0 to 1 in 1.0)
Freshwater Invertebrate (Indirect effects to DS)	5.69	Mortality Default Slope = 4.5 (2 – 9)	1 in 4.18E+08 (1 in 216 to 1 in 1.75E+31)	1 in 1.0 (1 in 1.07 to 1 in 1.0)
E/M Invertebrate (Indirect effects to DS	3.84	Mortality Default Slope = 4.5 (2 – 9)	1 in 4.18E+08 (1 in 216 to 1 in 1.75E+31)	1 in 1.0 (1 in 1.14 to 1 in 1.0)

In order to better characterize the potential risk to freshwater invertebrates, toxicity endpoints were also compared to exposure estimates based on aquatic dissipation studies and monitoring data collected in a rice growing area of California. These results are shown in Table 5-13.

Table 5-13. Acute and Chronic RQs for Aquatic Invertebrates Used to Evaluate Potential Indirect Effects to the CRLF and the DS Resulting from Potential Impacts to Food Supply. Exposure estimated based on aquatic dissipation studies and monitoring data.

Basis of Exposure Estimates	Peak EEC (µg/L)	Chronic EEC (µg/L)	Risk Quotients			
			FW Invertebrates		E/M Invertebrates[5]	
			Acute[1]	Chronic[2]	Acute[3]	Chronic[4]
Realistic Exposure Estimate in Paddy Water and Where Paddy Water is Released (CRLF and DS)						
Aquatic Dissipation Study (MRID 43404005)	438	202[5]	**4.33**	**202.00**	**2.92**	**63.13**
Aquatic Dissipation Study (Ross and Sava, 1986)	576	209[5]	**5.69**	**209.00**	**3.84**	**65.31**
Exposure Estimates Based on Measured Concentration in Rice Growing Areas of CA (CRLF and DS)						
Colusa Basin Drain on Sacramento River [6]	170[6]	5.37[7]	**1.68**	**5.37**	**1.13**	**1.68**

LOC exceedances (acute RQ $\geq$ 0.05; chronic RQ $\geq$ 1.0) are bolded. Acute RQ = peak EEC /96-hr LC_{50}; Chronic RQ = 21 day EEC/NOAEC; FW=freshwater; E/M=estuarine/marine

1- The freshwater invertebrate endpoint used to calculate acute RQs is the 96-hr LC_{50} of 101.2 µg/L for *Daphnia magna* (MRID 00025788).
2- The NOEAC used to calculate chronic RQs for freshwater invertebrates is 1 µg/L for the *Daphnia magna* based on reduced number of offspring produced (MRID 00079098).
3- The E/M acute endpoint used to calculate RQs is the 96-hr LC_{50} of 150 µg/L for the Mysid shrimp (MRID 00050667).
4- The E/M acute endpoint used to calculate RQs is the 21-day NOAEC of 3.2 µg/L for the Opossum shrimp based on reduced survival of offspring (MRID 43976801 or 40651314).
5- The chronic value represents the 21-day average concentration observed in the aquatic dissipation study.
6- The peak value represents the highest measured concentration in rice growing areas of California.
7- The chronic value represents the average of concentrations measured between May 9 and June 22, 2000 in the NAWQA monitoring.

More realistic RQs based on aquatic dissipation studies range from 4.33 – 5.69 (Table 5-2). At these RQs and using the default slope (4.5), the probability of an effect would still be approximately 1 in 1.0 (estimated using IEC version 1.1). The RQ (1.68) based on high concentrations measured in the Sacramento River before the 14-day holding period was put in place also exceed the LOC of 0.05.

Based on chronic exposure to freshwater invertebrates, the Agency's chronic risk LOC of 1 is exceeded for the concentrations estimated in the rice paddy (RQ = 968), when tail water is released from rice paddies after 14-days (RQ=3.46), concentrations measured in aquatic dissipation studies (RQs = 202 - 209), and for high concentrations measured in rice growing areas of California (RQ = 5.37). For E/M invertebrates, the LOC of 1.0 is also exceeded for all exposure scenarios with RQs ranging from 1.68 – 65.31 (Table 5-2).

The NOAEC (1.0 µg/L) used to calculate the chronic freshwater invertebrate RQs is based on an endpoint of number of offspring produced and the EECs are well above the NOAEC.
The LOAEC for this study was 3.0 µg/L. A reduction in the number of offspring has the

potential to influence the abundance of aquatic invertebrates and to decrease the number of prey items available for the CRLF and DS. Therefore, use of thiobencarb on rice is likely to adversely affect CRLF and DS due to decreased aquatic invertebrate prey items.

DS (Estuarine/marine aquatic invertebrates)

The DS eats small zooplankton. They primarily eat planktonic copepods, cladocerans, amphipods, and insect larvae. However, the most important food organism appears to be *Eurytemora affinis,* which is a euryhaline copepod (USFWS, 1996, 2004b). No estuarine/marine studies were submitted examining toxicity of thiobencarb to copepods. A supplemental field study is available from the open literature for a non-native freshwater *cyclopoida* and *calanoida* (E62293). In this field study, conducted with a formulation thiobencarb, the 7-day LOAEL was 187.5 µg a.i./L (effects observed were emergence success). Risk quotients were calculated based on the most sensitive aquatic invertebrate endpoint, the mysid value of 150 µg/L (MRID 00050667). Based on toxicity data from mysid and residues measured in aquatic dissipation studies, the RQs for thiobencarb use on rice (RQs range from 2.92 – 3.84) exceed the Agency's acute risk LOCs (listed or non-listed species). RQs estimated using monitoring data range from <0.00 – 1.13 and also exceed LOCs. At the highest RQ (3.84) and using a default slope of 4.5, the probability of an effect would be approximately 1 in 1.00 (estimated using IEC version 1.1). Therefore, impacts to potential estuarine/marine invertebrate prey are expected from acute exposure to thiobencarb.

For chronic risk to E/M invertebrates, risk quotients were calculated based on the 21-day NOEAC of 3.2 µg a.i./L for the Opossum shrimp which is based on survival of offspring (MRID 43976801 or 40651314). The LOAEC for this study is 6.2 µg a.i./L. It is likely that exposure to water released from rice paddies could result in exposures estimated using aquatic dissipation data and measured in monitoring studies. These results indicate that effects to E/M aquatic invertebrates may occur near rice paddies. Risk would decrease as exposure decreased with distance from rice paddies. Based on the weight of evidence, indirect risk to DS due to reductions in E/M aquatic invertebrate prey is likely to occur with use of thiobencarb on rice in California.

5.2.2.c. Potential Impacts to Aquatic Plants

CRLF tadpoles consume primarily algae, and DS larvae consume phytoplankton. Algal RQs ranged from approximately 20.59 – 118.71 based on modeling data. RQs were calculated based on the 5-day EC_{50} of 17 µg/L for green algae (MRID 41690901). Vascular plant RQs were only exceeded when based on modeled data and inside rice paddies (RQs ranged from 0.45 – 2.62). These results indicate that there is a potential for effects on aquatic plants and indirect effects to CRLF and DS based on effects on food and habitat. In order to better characterize potential risk to aquatic plants, RQs were estimated based on exposure estimates from aquatic dissipation studies and monitoring data. These results are shown in Table 5-14.

Table 5-14. Summary of RQs for Vascular and Non-Vascular Aquatic Plants Using Exposure from Aquatic Dissipation Studies and Monitoring Data.

Basis of Exposure Estimates	Peak EEC (µg/L)	Risk Quotients	
		Non-vascular Plants[1]	Vascular Plants[2]
Realistic Exposure Estimate in Paddy Water and Where Paddy Water is Released (CRLF and DS)			
Aquatic Dissipation Study (MRID 43404005)	438	**25.76**	0.57
Aquatic Dissipation Study (Ross and Sava, 1986)	576	**33.88**	0.75
Exposure Estimates Based on the Measured Concentration in Rice Growing Areas of CA (CRLF and DS)			
Colusa Basin Drain on Sacramento River [3]	170[3]	**10.00**	0.22

LOC exceedances (RQ $\geq$ 1.0) are bolded. RQ = peak EEC /5-day EC50 for nonvascular plants or 14-day EC50 for vascular plants.

1- The non-vascular plant endpoint used to calculate RQs for the 5-day EC50 of 17 µg/L for the Green algae (MRID 41690901).

2- The vascular plant endpoint used to calculate acute RQs is the 14-day EC_{50} of 770 µg/L for Duckweed (MRID 41690901).

3- Represents the highest measured concentration in rice growing areas of California.

Aquatic dissipation studies and monitoring data support the conclusions drawn from the modeling data. Nonvascular plant RQs ranged from 10.00 – 33.88. Vascular plant RQs ranged from 0.22 – 0.75. LOCs for vascular plants were only exceeded based on exposure estimated in the rice paddy. LOCs for non-vascular plants were exceeded using EECs estimated for the rice paddy, based on measured concentrations in rice paddies, and based on measured concentrations in rice growing areas. Since the RQs are exceeded, there is a potential for indirect effects to the CRLF and DS that rely on non-vascular aquatic plants during at least some portion of their life-cycle.

5.2.2.d. Indirect Effects, Terrestrial-Phase CRLFs

As discussed in Section 2.5 and 2.6, the diet of terrestrial-phase CRLFs includes terrestrial invertebrates, small mammals, and amphibians. Potential impacts to each of these potential food items are evaluated below.

5.2.2.e. Terrestrial Invertebrates

When the terrestrial-phase CRLF reaches juvenile and adult stages, its diet is mainly composed of terrestrial invertebrates. Thiobencarb is classified as practically nontoxic to non-target insects on an acute contact exposure basis. An acute contact LD_{50} for terrestrial invertebrates could not be determined based on available data. At the highest concentration tested in the oral test, 23.3% of the bees died in the treatment group as opposed to 6.7 and 3.3% in the negative and solvent control groups, respectively. In the contact exposure group, 15% of bees died as compared to 6.7 and 10% in the control groups. The 48-hr LD_{50} is greater than 100 µg a.i./bee for both studies. No sublethal effects were observed. Only one concentration was used in this study; therefore, a definitive LD_{50} value and response slope could not be determined. Using an LD_{50} of >100 µg a.i./bee results in and EEC/highest dose tested ratio of 0.69 and 0.08 for small and large insects, respectively; however, it is not clear if the actual RQs are above or below the interim LOC of 0.05 for acute risk to endangered terrestrial invertebrates.

In the submitted contact honey bee study, a concentration of 781 µg/g bee resulted in 15% mortality. Based on T-REX, a flowable thiobencarb application of 4 lb a.i./acre results in EEC values of 60 and 540 ppm for large and small insects, respectively. Therefore, the concentration on the site of application at the maximum allowable application rate is not expected to reach levels high enough to cause 15% mortality in large insects. However, depending on the slope of the dose response curve, the EC_{50} value could be slightly above 781 µg/g bee and could result in an RQ greater than the listed species LOC of 0.05. The likelihood of risk to terrestrial invertebrates is low; however, risk cannot be precluded without a definitive toxicity endpoint.

5.2.2.f. Mammals

Terrestrial-phase CRLFs consume small mammals. This assessment used a 15 g mammal as a potential mammalian prey. For thiobencarb formulations applied as a broadcast spray, RQs for mammals consuming short grass, tall grass, and broadleaf plants/small insects exceed the Agency's acute risk to listed species LOC (RQs range from 0.18 – 0.40). Using the highest RQ of 0.40 and the default slope of 4.5, the probability of individual effect is approximately 1 in 2.73 (estimated using IEC version 1.1). For granular formulations, the RQ (1.22) also exceeds the listed species LOC of 0.05 based on an $LD50/ft^2$ analysis. Assuming a default probit slope of 4.5, the probability of an individual effect would be approximately 1 in 1.54 (estimated using IEC version 1.1). Probit analysis for mammals are summarized in Table 5-15.

Table 5-15. Probit Dose-Response Analysis for Mammals and Potential Indirect Effects to CRLF

Species Represented	Maximum RQ	Acute Effect Slope (95% C.I.)	Chance of Individual Effect at Listed Species LOC (95% C.I.)	Chance of Individual Effect at Derived Acute RQ[1] (95% C.I.)
Mammals (liquid formulations)	0.40	Mortality Default Slope = 4.5 (2 – 9)	1 in 2.94E+05 (1 in 44 to 1 in 8.86E+18)	1 in 1.0 (1 in 4.69 to 1 in 5850)
Mammals (granular formulations)	1.22	Mortality Default Slope = 4.5 (2 – 9)	1 in 2.94E+05 (1 in 44 to 1 in 8.86E+18)	1 in 1.54 (1 in 1.76 to 1 in 1.28)

Regarding the potential for impacts from chronic exposure, all of the RQs for chronic exposure exceed the Agency's chronic risk LOC. The RQs range from 3.00 – 416.45. These RQs are based on a NOAEC of 1 mg/kg-diet from a 2-generation study (MRID 40446201) on the Fischer 344 rat. Decreased body weight gain, food consumption, and feeding efficiency were observed at 100 mg/kg/day. No reproductive effects were observed in the mammalian multigeneration studies at 100 mg/kg/day (MRID 40446201). It is uncertain whether a decrease in body weight gain would result in effects in survival and reproduction. The prenatal developmental studies in the rat and rabbit support the use of the body weight endpoint for the effects determination. In a developmental study in the rat an increase in skeletal anomaly and increased number of runts were observed at 150 mg/kg/day (NOAEL = 25 mg/kg/day; MRID 00115248). A decreased body weight gain was also observed in that study. Due to the uncertainty associated with possible effects based on an endpoint of body weight alone, an EEC/reproductive NOAEL of ≥100 mg/kg/day were calculated. Chronic dose-based ratios range from <0.53 – <83.29. Ratios could exceed 1.0 for mammals consuming short grass, tall grass, broadleaf plants/small insects, and fruits/pods/seeds/large insects. Chronic dietary-based ratios range from <0.60 – <9.60 and could exceed the LOC of 1.0 for short grass, tall grass, broadleaf plants/small insects, and fruits/pods/seeds/large insects. It is uncertain whether actual RQs would exceed the LOCs as possible exposure exceeds the highest dose tested in the two-generation study. Based on effects observed on body weight, in the developmental studies, and on the reproductive effects observed in birds, chronic risk to mammals cannot be discounted. Therefore, the Agency concludes that thiobencarb use on rice has the potential to impact mammalian prey populations and thus reduce prey of the CRLF.

5.2.2.g. Amphibians

CRLF are known to prey on aquatic-phase amphibians. As discussed in Section 5.2.1.a, direct effects to aquatic phase amphibians are likely. Therefore, indirect effects to CRLF from a decline in potential aquatic phase amphibian prey are expected from the use of thiobencarb on rice in California.

Terrestrial amphibian prey of the CRLF include small amphibians such as tree frogs that do not prey on mammals. Therefore, the mammalian food group is not relevant in the evaluation of potential reductions in amphibian prey abundance. As discussed in Section 5.2.1.b, direct effects to terrestrial phase amphibians are likely. Therefore, indirect effects to CRLF from a decline in potential terrestrial amphibian prey are expected from the use of thiobencarb on rice in California.

5.2.2.h. Potential Effects to 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, such as fish and frogs. Emergent plants help reduce sediment loading and provide stability to nearshore areas and lower streambanks. In addition, vascular aquatic plants are important as attachment sites for egg masses of aquatic species. Results of the indirect effects assessment are used as the basis for the habitat modification analysis. As discussed in Section 5.2.2.c, aquatic plants are likely to be impacted from the use of thiobencarb on rice. Therefore, impacts to aquatic plants found near thiobencarb use sites are expected.

Terrestrial plants serve several important habitat-related functions for the listed assessed species. Among other things, 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 watershed, and serving as an energy source (CRLF and DS).

In addition to providing shelter and cover from predators while foraging, upland vegetation, including grassland and woodlands, provides cover during dispersal (CRLF).

Based on the results of the submitted terrestrial plant toxicity studies and the reported terrestrial plant incidents, the herbicide thiobencarb is phytotoxic to many plant species (seedling emergence endpoints are more sensitive than vegetative vigor endpoints). Additionally, monocots are more sensitive to thiobencarb than dicots, based on available data. Only dicots exposed to spray drift from aerial applictions of liquid formations exceed the LOC of 1.0. Spray drift EECs for granular formulations are assumed to be low because spray drift is expected to be minimal. Exposure to runoff from rice paddies is not evaluated as runoff is not expected to commonly occur in rice paddies.

A general conclusion that can be drawn from these data is that the inhibition of new growth may occur in non-target terrestrial plants from registered uses of thiobencarb. Inhibition of new growth could result in degradation of high quality riparian habitat over time because as older growth dies from natural or anthropogenic causes, plant biomass may be prevented from being replenished in the riparian area. Inhibition of new growth may also slow the recovery of degraded riparian areas that function poorly due to sparse vegetation because thiobencarb deposition onto bare soil would be expected to inhibit the growth of new vegetation. Additionally, because effects were seen in most species tested in the seedling emergence and vegetative vigor studies, it is likely that many species of herbaceous plants could be potentially affected by exposure to thiobencarb. This is supported by the seven plant incidence where effects on rice and an unspecified plant were reported after use of thiobencarb.

It is difficult to estimate the magnitude of potential impacts of thiobencarb use on riparian habitat and the magnitude of potential effects on stream water quality from such impacts as they relate to survival, growth, and reproduction of the CRLF and DS. The level of exposure and any resulting magnitude of effect on riparian vegetation are expected to be highly variable and dependent on many factors. The extent of runoff and/or drift into stream corridor areas is affected by the distance the thiobencarb use site is offset from the stream, local geography, weather conditions, and quality of the riparian buffer itself. The sensitivity of the riparian vegetation is dependent on the susceptibility of the plant species exposed to thiobencarb and composition of the riparian zone (*e.g.*, vegetation density, species richness, height of vegetation, width of riparian area).

In summary, terrestrial and aquatic plant RQs are above plant LOCs; therefore, labeled use of thiobencarb has the potential to affect both aquatic and riparian vegetation within CRLF and DS habitats.

5.2.3. Spatial Extent of Potential Effects

Since LOCs are exceeded, analysis of the spatial extent of potential effects is needed to determine where effects may occur in relation to the treated site. If the potential area of usage and subsequent effects overlaps with CRLF and DS critical habitat or areas of occurrence, a likely to adversely affect determination is made. If the area of potential effects and the CRLF and/or DS critical habitat and areas of occurrence do not overlap, a no effect determination is made.

To determine this area, the footprint of thiobencarb's use pattern is identified, using corresponding land cover data. This area is defined by the cultivated crop land cover based on potential use on rice. Actual usage is expected to occur in a smaller area as rice is only grown in a portion of the cultivated crop land cover class. The spatial extent of the effects determination also includes areas beyond the initial area of concern that may be impacted by runoff and/or spray drift (potential use areas + distance down stream or down wind from use sites where organisms relevant to the CRLF and/or DS may be affected). The determination of the buffer distance and downstream dilution for spatial extent of the effects determination is described below.

5.2.3.a. Spray Drift

In order to determine terrestrial and aquatic habitats of concern due to thiobencarb exposures through spray drift, it is necessary to estimate the distance that spray applications can drift from the treated area and still be present at concentrations that exceed levels of concern. Applications of thiobencarb granular formulations are not expected to result in any spray drift. For the flowable uses, a quantitative analysis of spray drift distances was completed using AgDRIFT (v. 2.01) using default inputs for ground applications (*i.e.*, high boom, ASAE droplet size distribution = Very Fine to Fine, 90^{th} data percentile) and aerial applications (*i.e.*, ASAE Very Fine to Fine).

Terrestrial Spray Drift Distances

Direct Effects. For direct effects to the terrestrial-phase CRLF, the highest RQs for birds that eat small insects were used to estimate the fraction of the application rates that would no longer exceed the listed species LOC (*i.e.*, fraction of applied = LOC/RQ) for both acute and chronic RQs. This number was used in AgDRIFT to calculate the distance from the field where the amount of thiobencarb that equaled the 'fraction of applied' would be expected to occur (as spray drift) (Table 5-16). The terrestrial spray drift distance needed to reduce exposure to levels below LOCs for direct effects to the CRLF is 272 feet for aerial applications of liquid formulations and 26 feet for ground applications.

Indirect Effects. For indirect effects, a spray drift analysis is conducted using endpoints used to evaluate indirect effects. The most sensitive species were terrestrial plants for terrestrial phase CRLF. The distance from the field for terrestrial plants is based on the most sensitive terrestrial plant non-listed species endpoint (*i.e.*, monocot seedling emergence EC_{25} = 0.0019 lb a.i./acre). This endpoint is used to estimate the fraction of the application rates that would no longer exceed the 25% level of effects for terrestrial plants (*i.e.*, fraction of applied = 0.0019 lb a.i./acre divided by the application rate in lb a.i./acre) (Table 5-16). The terrestrial spray drift distance to result in no LOC exceedances for the terrestrial-phase CRLF is greater than 1000 feet for aerial applications and 433 feet for ground applications.

Table 5-16. Distance from Thiobencarb Use Site Needed to Reduce Spray Drift to Levels that Do Not Exceed Terrestrial Acute and Chronic LOCs for Direct and Indirect Effects (Point Deposition Estimate)

Endpoint or Taxa Evaluated	Type of Endpoint	RQ Range	Fraction of Applied = LOC/RQ*	Spray Drift Distance to Result No Effect (feet)	
				Aerial Applications	Ground Applications
Direct Effects to CRLF					
Direct Effects to Terrestrial Phase CRLF (surrogate bird)	Acute: No mortality	<0.00 - <0.18	<1.80	n/a	n/a
	Chronic: Reproduction	0.19 - 9.15	0.11	**272**	**26**
Indirect Effects to Terrestrial Phase CRLF					
Mammals	Acute: mortality	0.01 - 0.40	0.25	85.3	13.12
	Chronic: Body weight	3.00 416.45	0.0002	> 1000	> 1000
Terrestrial invertebrates	15% mortality	<0.08 - <0.69	n/a	n/a	n/a
Non-listed Dicot terrestrial plants	Seedling Emergence Mortality	<0.1 - 12.20	0.0205	> 1000	121
Non-listed Monocot terrestrial plants	Seedling Emergence Shoot Length	<0.1 - 52.63	0.00475	**> 1000**	**433**

Bold values show the maximum spray drift distances needed to reduce exposure to below levels expected to result in LOC exceedances.

*For terrestrial plants, Fraction of applied is the lowest EC_{25}/maximum application rate in lbs a.i./A. Endpoints used in the calculation were the monocots - EC_{25} = 0.019 lb a.i./acre (seedling emergence) and dicots - EC_{25} = 0.082 lbs a.i./acre [seedling emergence, cabbage)] (MRID 41690902).

Aquatic Spray Drift Distances

Direct Effects. For direct effects to aquatic-phase CRLF and DS, the distance from the site of application in which spray drift could reach levels high enough to exceed the acute risk to endangered species LOC, the 'active rate' (*i.e.*, the highest maximum labeled rate) and the 'initial average concentration' (*i.e.*, Toxicity Endpoint × LOC) were used as input into AgDRIFT. For this analysis, the farm pond (*i.e.*, a pond with a depth of 2 meters and a downwind width of 63.61 m and flight line width of 157.21 m) was used as a proxy for CRLF and DS habitat. The other AgDRIFT inputs were the same as described above in the terrestrial distance analysis. The results of these calculations are shown in Table 5-17. The aquatic spray drift distance needed to reduce exposure to levels below LOCs for direct effects for CRLF is greater than 1000 feet for aerial applications and 207 feet for ground applications of liquid formulations. The aquatic spray drift distance needed to reduce exposure to levels below LOCs for direct effects to the DS is greater than 1000 feet for aerial applications and 351 feet for ground applications.

Table 5-17. Distance from Thiobencarb Use Site Needed to Reduce Spray Drift to Levels that Do Not Exceed Acute and Chronic LOCs in Aquatic Environments

Endpoint or Taxa Evaluated	Type of Endpoint	RQ Range	Endpoint Used to Evaluate Risk (µg/L)	Initial Average Concentration = Toxicity Endpoint ×LOC	Spray Drift Distance to Result in No Effect (feet)	
					Aerial App	Ground App
Direct Effects to Aquatic Phase CRLF and DS						
Freshwater fish in Paddy	Acute: mortality	0.39 - 4.59	440	22	> 1000	151
	Chronic: Posthatch survival	0.26 - 46.10	21	21	**> 1000**	**207**
Freshwater fish in Tailwater	Acute: mortality	0.39 - 1.31	440	22		
	Chronic: Posthatch survival	0.26 - 3.81	21	21		
Estuarine/Marine fish	Acute: mortality	0.83 - 2.82	204	10.2	**> 1000**	**351**
	Chronic: wet weight	0.90 - 13.33	6	6	> 1000	79
Indirect Effects to Aquatic Phase CRLF and DS						
Freshwater invertebrates	Acute: mortality	1.68 – 19.94	101.2	5.06	> 1000	39
	Chronic: Offspring produced	5.37 – 968	**1**	**1**	**> 1000**	**358**
Estuarine/Marine Invertebrates	Acute: mortality	2.92 – 3.84	150	7.5	> 1000	16
	Chronic: Posthatch survival	20.61 – 21.33	3.2	3.2	> 1000	89
Non-listed Aquatic vascular plants	Frond production	0.22-2.62	770	770	0	0
Non-listed Aquatic non-vascular plants	Cell Density/Inhibition of Growth	0.04-118.71	17	17	> 1000	207

App. = Application
Bold values show the maximum spray drift distances needed to reduce exposure to below levels expected to result in LOC exceedances.

Indirect Effects. For indirect effects, a spray drift analysis is conducted using endpoints used to evaluate indirect effects. The most sensitive species were freshwater invertebrates for the aquatic phase CRLF and DS. For freshwater invertebrates, the most sensitive 21-day NOAEC of 1.0 µg/L for the daphnia was used to estimate the maximum aquatic spray drift distance by calculating the initial average concentration (Toxicity endpoint x LOC) for input into AgDrift. This number is used in AgDrift to calculate the distance from the field where the amount of thiobencarb that equals the 'initial average concentration' would be expected to occur (as spray drift) (Table 5-17). The aquatic spray drift distance to result in no LOC exceedances for the aquatic-phase CRLF and DS is greater than 1000 feet for aerial applications and 358 feet for ground applications (see Table 5-17).

Looking at all estimated spray drift distances indicates that effects may occur at greater than 1000 feet for both aerial and ground applications. Because the LOCs are expected to be exceeded at greater than 1000 feet from the use site, it is not known where risk would fall below LOCs for aerial and ground applications.

5.2.3.b. Downstream Dilution Analysis

The downstream extent of exposure in streams and rivers where the EEC could potentially be above levels that would exceed the most sensitive LOC is calculated using the downstream dilution model. To complete this assessment, the greatest ratio of aquatic RQ to LOC was estimated. Using an assumption of uniform runoff across the landscape, it is assumed that streams flowing through treated areas (*i.e.*, the initial area of concern) are represented by the modeled EECs; as those waters move downstream, it is assumed that the influx of non-impacted water will dilute the concentrations of thiobencarb present. The highest RQ/LOC ratio and the land cover class (cultivated crops) are used as inputs into the downstream dilution model.

Using a 48-hr LC_{50} value of 101.2 µg/L for *Daphnia magna*, an LOC of 0.05, and a maximum peak EEC of 2018 µg/L from the Tier I Rice Model yields an RQ/LOC ratio of 399 (101.2/0.05). The downstream dilution approach is described in more detail in Appendix N. This value has been input into the downstream dilution model and results in a distance of 285 kilometers which represents the maximum continuous distance of downstream dilution from the edge of the initial area of concern where LOCs may be exceeded in the aquatic environment.

5.2.3.c. Overlap of Potential Areas of Effect and CRLF and DS Habitat

The spray drift and downstream dilution analyses help to identify areas of potential effect to the CRLF and DS from registered uses of thiobencarb. The potential area of effects for the CRLF and DS from thiobencarb spray drift extend from the site of application to greater than 1000 feet from the site of application. For exposure to runoff and spray drift, the area of potential effects extends up to 285 km downstream from the site of application. When these distances are added to the footprint of the initial area of concern (which represents potential thiobencarb use sites) and compared to CRLF and DS habitat, there are several areas of overlap (Figure 5-1 and Figure 5-2**)**. The overlap between the areas of effect and CRLF and DS habitat, including designated critical habitat, indicates that thiobencarb use in California has the potential to affect the CRLF and DS.

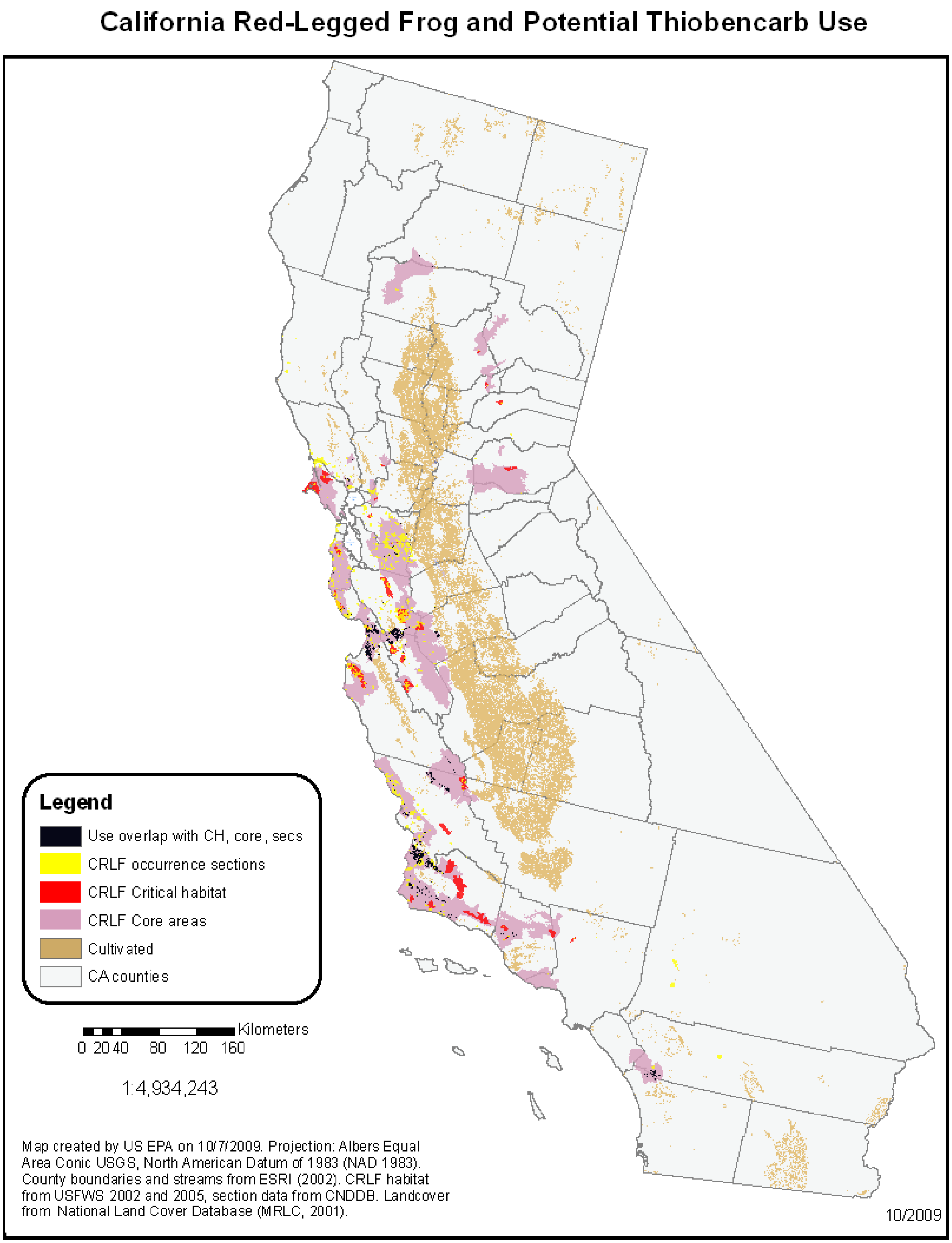

Figure 5-1. Map Showing the Overlap of CRLF Critical Habitat, Occurrence Sections, and Core Areas with the NLCD Cultivated Crop Land Cover Class

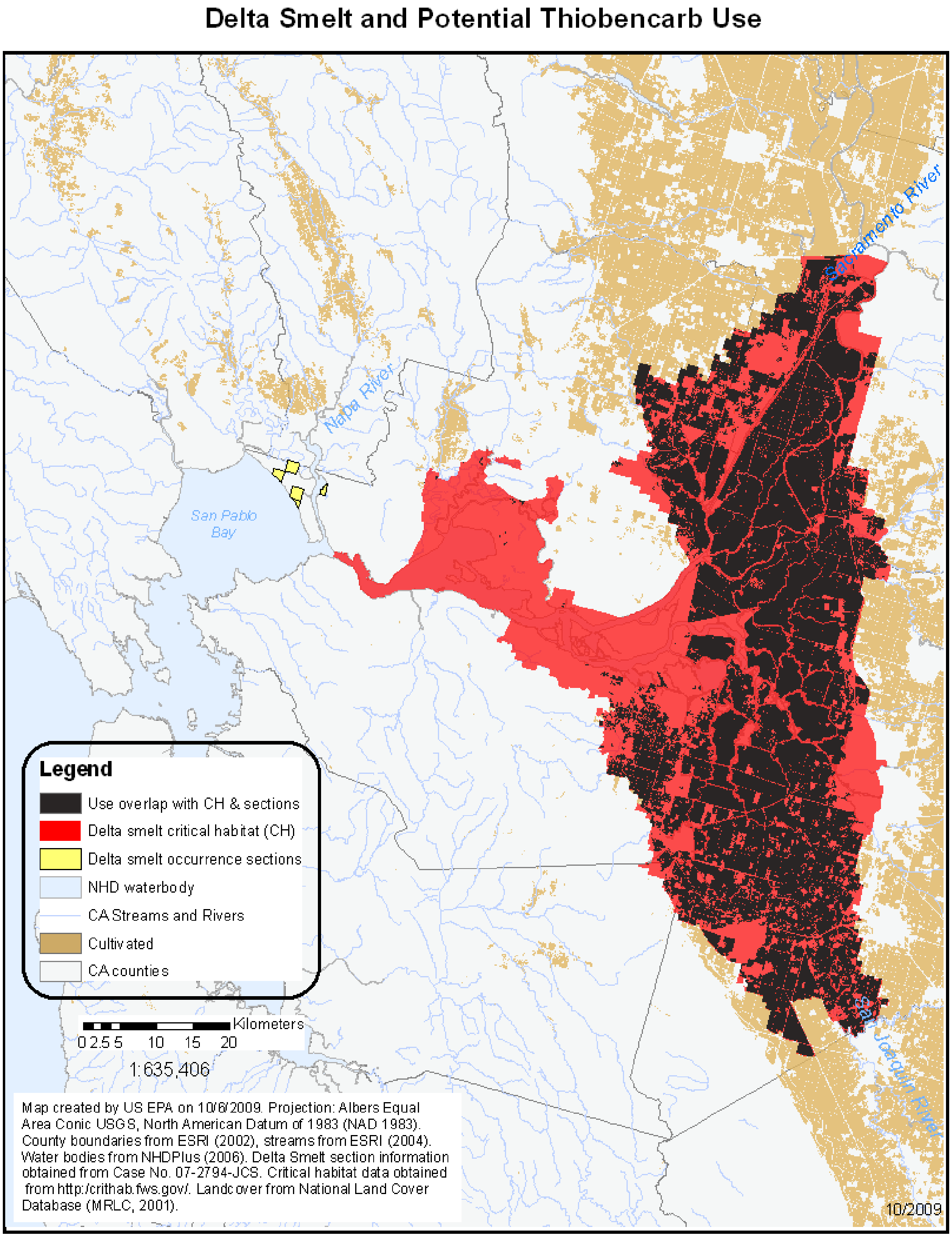

Figure 5-2. Map Showing the Overlap of DS Critical Habitat and Occurrence Sections Identified by Case No. 07-2794-JCS with the NLCD Cultivated Crop Land Cover Class.

5.3. Effects to Designated Critical Habitat

The risk conclusions for the designated critical habitat are based on conclusions described for indirect effects previously described. Potential effects to habitat are described below.

5.3.1. CRLF Habitat Modification Analysis

5.3.1.a. Aquatic-Phase PCEs

Three of the four assessment endpoints for the aquatic-phase primary constituent elements (PCEs) of designated critical habitat for the CRLF are related to potential effects to aquatic and/or terrestrial plants:

- Alteration of channel/pond morphology or geometry and/or increase in sediment deposition within the stream channel or pond: aquatic habitat (including riparian vegetation) provides for shelter, foraging, predator avoidance, and aquatic dispersal for juvenile and adult CRLFs.
- Alteration in water chemistry/quality including temperature, turbidity, and oxygen content necessary for normal growth and viability of juvenile and adult CRLFs and their food source.
- Reduction and/or modification of aquatic-based food sources for pre-metamorphs (*e.g.,* algae).

Conclusions for potential indirect effects to the CRLF via direct effects to aquatic and terrestrial plants are used to determine whether effects to critical habitat may occur. Additionally, direct effects to aquatic and riparian plants could result in indirect effects to aquatic invertebrates and other aquatic vertebrates other than the CRLF. As previously discussed, thiobencarb may cause effects to habitat by potentially impacting aquatic plants and terrestrial plants.

The remaining aquatic-phase PCE is "alteration of other chemical characteristics necessary for normal growth and viability of CRLFs and their food source." Thiobencarb may impact algae as food items for tadpoles. Thiobencarb may also impact riparian areas that are predominantly grassy or herbaceous, and the potential areas of effect overlap with designated critical habitat for the CRLF (see Figure 5-1). Therefore, there is a potential for effects to habitat by potentially impacting the chemical characteristics of the habitat.

5.3.1.b. Terrestrial-Phase PCEs

Two of the four assessment endpoints for the terrestrial-phase PCEs of designated critical habitat for the CRLF are related to potential effects to terrestrial plants:

- Elimination and/or disturbance of upland habitat; ability of habitat to support food source of CRLFs: Upland areas within 200 ft (0.06 km) of the edge of the riparian vegetation or drip line surrounding aquatic and riparian habitat that are comprised of grasslands, woodlands, and/or wetland/riparian plant species that provides the CRLF shelter, forage, and predator avoidance.

- Elimination and/or disturbance of dispersal habitat: Upland or riparian dispersal habitat within designated units and between occupied locations within 0.7 mi (1.1 km) of each other that allow for movement between sites including both natural and altered sites which do not contain barriers to dispersal.

As an herbicide, thiobencarb may affect sensitive terrestrial plants; information from the reported terrestrial plant incident data support this. Additionally, terrestrial plant LOCs are exceeded for use of thiobencarb on rice and the potential areas of effect overlap with designated critical habitat for the CRLF (see Figure 5-1).

The third terrestrial-phase PCE is "reduction and/or modification of food sources for terrestrial-phase juveniles and adults." To assess the impact of thiobencarb on this PCE, acute and chronic toxicity endpoints for terrestrial invertebrates, mammals, and terrestrial-phase frogs are used as measures of effects. There is a potential for habitat modification based on potential reductions in prey base (mammals and frogs, as previously described), and, again, the areas of potential effect overlap with CRLF critical habitat (Figure 5-1).

The fourth terrestrial-phase PCE is based on alteration of chemical characteristics necessary for normal growth and viability of juvenile and adult CRLFs and their food source. There is a potential for habitat modification based on potential direct (Section 5.2.1) and indirect effects (Sections 5.2.2) to terrestrial-phase CRLFs.

5.3.2. Delta Smelt Habitat Modification Analysis

Primary constituent elements (PCEs) of designated critical habitat for the DS include the following:

- 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).

- 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 two parts per thousand 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.

- PCEs also include 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.

The potential for direct effects to the DS from thiobencarb use could not be precluded based on incident data. Furthermore, it was concluded that thiobencarb is likely to adversely affect the DS by potentially affecting its habitat (aquatic and terrestrial plants) and the potential areas of effect overlap with critical habitat designated for DS (Figure 5-2). Finally, monitoring data in the Sacramento river indicate that exposure may be high enough to result in effects on aquatic organisms. Therefore, thiobencarb may also affect critical habitat of the DS that is located in close proximity to thiobencarb use sites.

5.4. Effects Determinations

5.4.1. CRLF

The weight of evidence indicates that thiobencarb use has the potential to directly adversely affect CRLF. Acute and chronic risk to aquatic-phase CRLF is high based on the RQ analyses. Although the acute risk to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low, risks could not be precluded at this time. The potential chronic risk to terrestrial-phase CRLF from chronic dietary exposure cannot be precluded and exists for amphibians consuming small herbivore mammals and small insects.

Regarding the potential for indirect effects, exceedance of all acute and chronic LOCs indicates that effects to aquatic invertebrates are likely to occur. While the available data for terrestrial invertebrates indicate that risk to terrestrial invertebrates is low, it is possible that effects to terrestrial invertebrate LOCs could be exceeded if the EC_{50} was slightly higher than the highest dose tested here 15% mortality occurred. Impacts to non-vascular aquatic and terrestrial plants, however, are expected from use of thiobencarb. Additionally, the Agency concludes that there exists the potential, which cannot currently be precluded, for thiobencarb use to impact amphibian prey populations to levels high enough to impact CRLF. Spatial analyses show that potential areas of effect from thiobencarb use overlap with CRLF habitat and their designated critical habitat.

Monitoring data, reported incidence, and usage data support the potential for thiobencarb to result in direct and indirect effects to the CRLF. Monitoring studies indicate that concentrations are likely to be high enough to result in toxicity to aquatic organisms. Monitoring data are shown to be highest in May and June, a young juvenile life stage of the CRLF (Orlando and

Kuivila, 2004). Three incidence were reported on terrestrial plant species (rice) for registered uses and the certainty that the use of thiobencarb caused the incidence was probable (see Section 4.7). Although aquatic field studies cannot conclusively show that exposure to thiobencarb resulted in declines to fish, aquatic invertebrates, and gravid shrimp, they do show that these endpoints were affected in areas where thiobencarb was applied (see Section 4.7.1). Based on the CADPR PUR data, from 1999 to 2006 an average of 308, 491 – 1,006, 327 lbs of thiobencarb per year were applied to rice in California. Usage of thiobencarb on rice in California is expected to occur and thiobencarb is often used at the maximum application rate (Table 2-6).

Therefore, the Agency makes a **"may affect, and likely to adversely affect**" determination for the CRLF and a **habitat modification determination** for their designated critical habitat based on the potential for direct and indirect effects and effects to the PCEs of critical habitat.

5.4.2. Delta Smelt

The weight of evidence indicates that thiobencarb use will directly adversely affect DS. Acute and chronic RQs exceed LOCs in rice growing areas and DS may be found in those areas during spawning. Regarding the potential for indirect effects, the impact from thiobencarb use to estuarine/marine invertebrate populations are also expected to be large enough to impact the DS indirectly. RQs were calculated based on a toxicity endpoint for *Daphnia magna* of 101.2 µg/L (MRID 00025788). Copepods are important in the diet of DS. A supplemental field study suggests that freshwater copepods may have a similar sensitivity to thiobencarb as *Daphnia magna* (LOAEC of 187.5 µg/L) (E62293). Impacts to non-vascular and vascular aquatic and terrestrial plants are expected from use of thiobencarb on rice. Spatial analyses show that potential areas of effect from thiobencarb use overlap with DS habitat and their designated critical habitat.

As discussed for the CRLF, monitoring data, reported incidence, and usage data support the potential for thiobencarb to result in direct and indirect effects to the CRLF. Monitoring data are shown to be highest in May and June, a time when DS may move into freshwater habitat to spawn (Orlando and Kuivila, 2004).

Therefore, the Agency makes a **"may affect, and likely to adversely affect**" determination for the CRLF and a **habitat modification determination** for their designated critical habitat based on the potential for direct and indirect effects and effects to the PCEs of critical habitat.

5.4.3. 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, none of the hypotheses can be rejected, meaning that the stated hypotheses represent concerns in terms of direct and indirect effects of thiobencarb on the CRLF and its designated critical habitat.

The labeled use of thiobencarb may:

- … directly affect terrestrial-phase CRLF by causing acute mortality or by adversely affecting chronic growth or fecundity;
- … indirectly affect the CRLF and the DS and/or affect their designated critical habitat by reducing or changing the composition of the food supply;
- … indirectly affect the CRLF and the DS and/or affect 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 the CRLF and the DS and affect their designated critical habitat by reducing or changing the composition of the terrestrial plant community in the species' current range;
- … indirectly affect the CRLF and the DS and affect 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).

6. Uncertainties

6.1. Exposure Assessment Uncertainties

6.1.1. Maximum Use Scenario

The screening-level risk assessment focuses on characterizing potential ecological risks resulting from a maximum use scenario, which is determined from labeled statements of maximum application rate and number of applications with the shortest time interval between applications. The frequency at which actual uses approach this maximum use scenario may be dependant on pest resistance, timing of applications, cultural practices, and market forces.

6.1.2. Aquatic Exposure Modeling of Thiobencarb

The Tier 1 Rice Model estimates only the concentration in the paddy water on the day of application, and does not account for any dissipation processes except sorption to sediment. Thus, it is likely to overestimate concentrations at later times, and thus to overestimate chronic exposure. The extended Tier 1 model uses the result of a single aquatic field dissipation study to estimate a lumped dissipation/degradation parameter, which may over- or under-estimate the rate of dissipation which would be derived from a larger number of field studies.

There are only two aquatic field dissipation studies for wet-seeded rice. Both studies were conducted in California, and only one measured degradates. It is not known how well these studies represent dissipation of thiobencarb in wet-seeded rice in other parts of the U.S.

Monitoring data from several states suggest that thiobencarb concentrations in open waters (not rice paddies) are in the low ppb range. However, the proximity of many monitoring sites to thiobencarb use sites (in time and space) is not known at this time. The best-characterized monitoring data is from California. The most highly contaminated sampling point, Colusa Basin Drain #5, is known to be surrounded by rice fields, and the sampling was done during the time of year when thiobencarb was being discharged from rice fields. The highest concentration observed at this location (37.4 ppb in 1994), suggests that water management practices are just adequate to meet the California public health goal of 70 ppb.

There is one field dissipation study in dry-seeded rice. It suggests that thiobencarb concentrations in flood water are low in comparison to wet-seeded fields. It is not known whether additional studies would yield higher or lower concentrations. Thus, the result of this study must be taken as a lower bound for dry-seeded rice.

Further degradation and dilution is expected to occur between the many areas where the DS are commonly found throughout much of the year.

The Agency acknowledges that there are some unique aquatic habitats that are not accurately captured by this modeling scenario and modeling results may, therefore, under- or over-estimate exposure, depending on a number of variables. For example, some organisms may inhabit water bodies of different size and depth and/or are located adjacent to larger or smaller drainage areas than that used in the Tier I Rice Model.

6.1.3. Uncertainties regarding dilution and chemical transformations in estuaries

Tier I Rice modeled EECs are intended to represent exposure of aquatic organisms in relatively small water bodies. Therefore it is likely that modeled EECs 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’ rice paddy. 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 (Fernandez *et al.*, 2005; Parikh *et al.*, 2004; 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. In general, model results are 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. Monitoring data do indicate that concentrations in estuaries are lower than those observed in upstream monitoring stations (see Section 3.1.1). 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 modeling results, and may, therefore, under- or over-estimate exposure, depending on the aforementioned variables.

Uncertainties associated with each of these individual components add to the overall uncertainty of the modeled concentrations. Additionally, model inputs from the environmental fate degradation studies are chosen to represent the upper confidence bound on the mean values that are not expected to be exceeded in the environment approximately 90 percent of the time. Mobility input values are chosen to be representative of conditions in the environment. The natural variation in soils adds to the uncertainty of modeled values. Factors such as application date, crop emergence date, and canopy cover can also affect estimated concentrations, adding to the uncertainty of modeled values. Factors within the ambient environment such as soil temperatures, sunlight intensity, antecedent soil moisture, and surface water temperatures can cause actual aquatic concentrations to differ for the modeled values.

6.1.4. Impact of Vegetative Setbacks on Runoff

Unlike spray drift, tools are currently not available to evaluate the effectiveness of a vegetative setback on runoff and loadings. The effectiveness of vegetative setbacks is highly dependent on the condition of the vegetative strip. For example, a well-established, healthy vegetative setback can be a very effective means of reducing runoff and erosion from agricultural fields. Alternatively, a setback of poor vegetative quality or a setback that is channelized can be ineffective at reducing loadings. Until such time as a quantitative method to estimate the effect of vegetative setbacks on various conditions on pesticide loadings becomes available, the aquatic exposure predictions are likely to overestimate exposure where healthy vegetative setbacks exist and underestimate exposure where poorly developed, channelized, or bare setbacks exist.

6.1.5. Exposure Resulting from Atmospheric Transport

Thiobencarb has been detected in precipitation samples in California. According to Suzuki *et al.* (2003), thiobencarb was detected in 59.8% of rainfall samples collected in Eastern Japan at a maximum concentration of 0.335 μg/L. These monitoring data are not expected to be representative of California; however, they are the only data available and can be used to determine whether further analysis of potential risk of thiobencarb in precipitation is necessary. Based on these data, it is possible that thiobencarb can be deposited on land in precipitation. Estimates of exposure of the CRLF, its prey and its habitat to thiobencarb included in this assessment are based only on transport of thiobencarb through spray drift from application sites. Current estimates of exposures of CRLF and its prey to thiobencarb through spray drift would be greater if consideration is given to deposition in precipitation. In the aquatic environment, the concentration in precipitation (0.335 µg/L) is a fraction of the aquatic EECs (968 - 2018 µg/L) predicted based on modeling and aquatic dissipation studies (EECs ranged from 70 – 438 µg/L. Rainwater could make a significant contribution to RQs estimated based on NAWQA and CDPR monitoring data where the representative EEC was 0.697; however, the other exposure estimates indicate that accounting for the contribution of rain to exposure is not necessary. Thiobencarb concentrations in rainwater are also a fraction of the terrestrial EECs (0.60 – 540 mg/kg-diet, for dietary exposure to birds and mammals) based on modeling. Therefore, precipitation is not expected to be a significant source of exposure. It was assumed that exposure due to the presence of thiobencarb in precipitation would have a minor impact on risk conclusions.

6.1.6. Potential Ground Water Contributions to Surface Water Chemical Concentrations

Although the potential impact of discharging ground water on CRLF populations is not explicitly delineated, it should be noted that, in some areas of the country, ground water could provide a source of pesticide to surface water bodies – especially low-order streams, headwaters, and ground water-fed pools. This is particularly likely if the chemical is persistent and mobile, the pesticide is applied to highly permeable soils overlying shallow unconfined ground water, and rainfall is sufficient to drive the chemical through the soil to ground water. Soluble chemicals that are primarily subject to photolytic degradation will be very likely to persist in ground water, and can be transportable over long distances. Similarly, many chemicals degrade slowly under anaerobic conditions (common in aquifers) and are thus more persistent in ground water. Under the right hydrologic conditions, this ground water may eventually be discharged to the surface – often supporting stream flow in the absence of rainfall. Continuously flowing low-order streams in particular are sustained by ground water discharge, which can constitute 100% of stream flow during baseflow (no runoff) conditions. Thus, it is important to keep in mind that pesticides in ground water may impact surface water quality during base flow conditions with subsequent impact on CRLF habitats. However, many smaller streams in CA are net dischargers of water to ground water that go dry during portions of the year and are not supplied by baseflow from ground water.

Although concentrations in a receiving water body resulting from ground water discharge cannot be explicitly quantified, it should be assumed that significant attenuation and retardation of the chemical will have occurred prior to discharge. Nevertheless, where thiobencarb is applied to highly permeable soils over shallow ground water where there is a net recharge to adjacent streams, ground water could still be a consistent source of chronic background concentrations in surface water, and may also add to surface runoff during storm events (as a result of enhanced ground water discharge typically characterized by the 'tailing limb' of a storm hydrograph).

6.1.7. Usage Uncertainties

County-level usage data were obtained from California's Department of Pesticide Regulation Pesticide Use Reporting (CDPR PUR) database. Eight years of data (1999 – 2006) were included in this analysis because statistical methodology for identifying outliers, in terms of area treated and pounds applied, was provided by CDPR for these years only. CDPR PUR documentation indicates that errors in the data may include the following: a misplaced decimal; incorrect measures, area treated, or units; and reports of diluted pesticide concentrations. In addition, it is possible that the data may contain reports for pesticide uses that have been cancelled. The CPDR PUR data does not include home owner applied pesticides; therefore, residential uses are not likely to be reported. As with all pesticide usage data, there may be instances of misuse and misreporting. The Agency made use of the most current, verifiable information; in cases where there were discrepancies, the most conservative information was used.

6.1.8. Terrestrial Exposure Modeling of Thiobencarb

The Agency relies on the work of Fletcher *et al.* (1994) for setting the assumed pesticide residues in wildlife dietary items. These residue assumptions are believed to reflect a realistic upper-bound residue estimate, although the degree to which this assumption reflects a specific percentile estimate is difficult to quantify. It is important to note that the field measurement efforts used to develop the Fletcher estimates of exposure involve highly varied sampling techniques. It is entirely possible that much of these data reflect residues averaged over entire above ground plants in the case of grass and forage sampling.

It was assumed that ingestion of food items in the field occurs at rates commensurate with those in the laboratory. Although the screening assessment process adjusts dry-weight estimates of food intake to reflect the increased mass in fresh-weight wildlife food intake estimates, it does not allow for gross energy differences. Direct comparison of a laboratory dietary concentration-based effects threshold to a fresh-weight pesticide residue estimate would result in an underestimation of field exposure by food consumption by a factor of 1.25 – 2.5 for most food items.

Differences in assimilative efficiency between laboratory and wild diets suggest that current screening assessment methods do not account for a potentially important aspect of food requirements. Depending upon species and dietary matrix, bird assimilation of wild diet energy ranges from 23 – 80%, and mammal's assimilation ranges from 41 – 85% (USEPA, 1993). If it is assumed that laboratory chow is formulated to maximize assimilative efficiency (*e.g.*, a value of 85%), a potential for underestimation of exposure may exist by assuming that consumption of food in the wild is comparable with consumption during laboratory testing. In the screening process, exposure may be underestimated because metabolic rates are not related to food consumption.

For the terrestrial exposure analysis of this risk assessment, a generic bird or mammal was assumed to occupy either the treated field or adjacent areas receiving a treatment rate on the field. Actual habitat requirements of any particular terrestrial species were not considered, and it was assumed that species occupy, exclusively and permanently, the modeled treatment area. Spray drift model predictions suggest that this assumption leads to an overestimation of exposure to species that do not occupy the treated field exclusively and permanently. Based on estimated exposure using AgDRIFT, TerrPlant may underestimate spray drift exposure for plants less than 200 feet from the edge of the field.

6.1.9. Spray Drift Modeling

Factors, including variations in topography, cover, and meteorological conditions over the transport distance are not accounted for by the AgDRIFT model (*i.e.*, it models spray drift from aerial and ground applications in a flat area with little to no ground cover and a steady, constant wind speed and direction). Therefore, in most cases, the drift estimates from AgDRIFT may overestimate exposure, especially as the distance increases from the site of application, since the model does not account for potential obstructions (*e.g.*, large hills, berms, buildings, trees, *etc*.). Furthermore, conservative assumptions are made regarding the droplet size distributions being modeled ('ASAE Very Fine to Fine') and boom height ('High') unless spray drift restrictions are specified on the label. Alterations in any of these inputs would decrease the area of potential effect.

6.2. Effects Assessment Uncertainties

6.2.1. Data Gaps and Uncertainties

<u>Avian Acute Toxicity for Passerine Species</u>

Data are typically required on one passerine species and either one waterfowl species or one upland game bird species for terrestrial, aquatic, forestry, and residential outdoor uses (USEPA, 2007). An acceptable acute oral toxicity study with Bobwhite quail and a subacute dietary study for the Mallard duck are available; however, data are not available for a passerine species. The proposed rule establishing this data requirement stated that the avian acute oral study would be required for outdoor uses because "…of concern in the scientific community that data from tests with mallards or quail may not always adequately characterize the risks that pesticides pose to songbirds. Recent evaluation of the data collected over the past 10 years indicates passerines are more sensitive to pesticides than larger birds such as mallards and quail (which are currently the recommended test species) (Ref. 2) and in 1996, the SAP supported the need for testing on passerines." (FR Vol. 70 No. 47; March 11, 2005; 12289). This indicates that the available avian endpoints are not good predictors of toxicity for passerine species and data are needed to adequately assess risk to songbirds. This could result in a misrepresentation of effects to the CRLF as avian endpoints are used as a surrogate to estimate direct effects to the CRLF.

<u>Avian Chronic Toxicity</u>

Only one acceptable chronic avian toxicity study is available. Other available chronic avian studies are missing information needed to fully evaluate and rely on the study results (Appendix I). Usually chronic studies on both waterfowl and upland game bird species are used to characterize chronic risk to avian species (USEPA, 2007). The limited data available to characterize chronic risk to avian species could result in an underestimation of direct effects to the CRLF.

Acute Fish Toxicity

The only acceptable fish acute toxicity study available on the TGAI is for the bluegill sunfish, a warmwater fish species. A study should also be available using the TGAI for a coldwater species. Additionally, not enough information is available to determine whether the TGAI or TEP is more toxic. Therefore, the most sensitive toxicity data for the TEP was used to calculate risk quotients. Data on a coldwater species using the TGAI would 1) reduce uncertainty on whether the formulation influenced toxicity of thiobencarb (*e.g.*, whether the TEP or TGAI was more toxic to fish) and 2) reduce uncertainty on possible underestimation of risk to coldwater fish species. Available open literature data indicate that coldwater species, white sturgeon and striped bass, are sensitive fish species (Appendix I). Available open literature data indicate that the value used to calculate risk quotients represents a sensitive species and provides a high end RQ value (see the species sensitivity distribution in Appendix I).

Chronic Fish Toxicity

A fish full life-cycle study was submitted for the fathead minnow. The study should have a corresponding acute study for the same species. No acute toxicity data were submitted for the fathead minnow. Additionally, the study only had two replicates. The low number of replicates and variability in the endpoints may have resulted in an inability to statistically detect differences between treatments and controls and thus may overestimate toxicity endpoints. Observed results suggested that if more replicates were available a difference may have been statistically significant between the control and 53 µg ai/L treatment group which would result in a higher RQ value. Chronic risk to fish and therefore, the CRLF and DS, may be underestimated.

Seedling Emergence

A NOAEL was not established for the seedling emergence terrestrial plant study (MRID 41690902). The test was classified as supplemental for the two most sensitive species, lettuce and ryegrass, because there was significant mortality of plants at the lowest test concentration. Additional testing should be completed for these two sensitive species using lower test concentrations that do not result in mortality of plants. The value added of this information is moderate. It would increase the confidence of the risk assessment on terrestrial plants. This data gap could result in an underestimation of indirect effects to the CRLF and DS.

6.2.2. Age Class and Sensitivity of Effects Thresholds

It is generally recognized that test organism age may have a significant impact on the observed sensitivity to a toxicant. The acute toxicity data for fish are collected on juvenile fish between 0.1 and 5 grams. Aquatic invertebrate acute testing is performed on recommended immature age classes (*e.g.*, first instar for daphnids, second instar for amphipods, stoneflies, mayflies, and third instar for midges).

Testing of juveniles may overestimate toxicity at older age classes for pesticide active ingredients that act directly without metabolic transformation because younger age classes may not have the enzymatic systems associated with detoxifying xenobiotics. In so far as the available toxicity data may provide ranges of sensitivity information with respect to age class, this assessment uses the most sensitive life-stage information as measures of effect for surrogate aquatic animals, and is therefore, considered as protective.

6.2.3. Impact of Multiple Stressors on the Effects Determination

The influence of length of exposure and concurrent environmental stressors to the CRLF and the DS (*i.e.*, construction of dams and locks, fragmentation of habitat, change in flow regimes, increased sedimentation, degradation of quantity and quality of water in the watersheds of the action area, predators, *etc*.) will likely affect the species' response to thiobencarb. Additional environmental stressors may increase sensitivity to the herbicide, although there is the possibility of additive/synergistic reactions. Timing, peak concentration, and duration of exposure are critical in terms of evaluating effects, and these factors are expected to vary both temporally and spatially within the action area. Overall, the effect of this variability may result in either an overestimation or underestimation of risk. However, as previously discussed, the Agency's LOCs are set to be protective given the wide range of possible uncertainties.

6.2.4. Use of Surrogate Species Effects Data

Freshwater fish are used as surrogate species for aquatic-phase amphibians. Some data are available on thiobencarb that evaluated its toxicity to amphibians. Overall, these data do not suggest that amphibians are more sensitive than fish to thiobencarb. Therefore, endpoints based on freshwater fish ecotoxicity data are assumed to be protective of potential direct effects to aquatic-phase amphibians including the CRLF, and extrapolation of the risk conclusions from the most sensitive tested species to the aquatic-phase CRLF is likely to overestimate the potential risks to those species. 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.

6.2.5. 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 purposes of defining the action area.

6.2.6. Exposure to Pesticide Mixtures

In accordance with the Overview Document and the Services Evaluation Memorandum (USEPA, 2004)(USFWS/NMFS/NOAA, 2004), this assessment considers the single active ingredient of thiobencarb. However, the assessed species and its environments may be exposed to multiple pesticides simultaneously. Interactions of other toxic agents with thiobencarb could result in additive effects, more than additive effects, or less than additive effects. As previously discussed, evaluation of pesticide mixtures is beyond the scope of this assessment because of the myriad of factors that cannot be quantified based on the available data. Those factors include identification of other possible co-contaminants where the CRLF and the DS reside and their concentrations, differences in the pattern and duration of exposure among contaminants, and the differential effects of other physical/chemical characteristics of the receiving waters (*e.g.* organic matter present in sediment and suspended water). Evaluation of factors that could influence additivity/synergism/antagonism is beyond the nature and quality of the available data to allow for an evaluation. However, it is acknowledged that not considering mixtures could over- or under-estimate risks depending on the type of interaction and factors discussed above.

6.2.7. Uncertainty in the Potential Effect to Riparian Vegetation vs. Water Quality Impacts

Effects to riparian vegetation were evaluated using submitted guideline seedling emergence and vegetative vigor studies. LOCs were exceeded for seedling emergence endpoints with the seedling emergence endpoint being considerably more sensitive than vegetative vigor endpoints. Based on LOC exceedances and the lack of readily available information to allow for characterization of riparian areas of the CRLF and the DS, it was concluded that thiobencarb use is likely to adversely affect these species by potentially impacting grassy/herbaceous riparian vegetation resulting in increased sedimentation. However, soil retention/sediment loading is dependent on a number of factors including land management and tillage practices. Use of herbicides (including thiobencarb) may be incorporated into a soil conservation plan. Therefore, although this assessment concludes that thiobencarb is likely to adversely affect the assessed listed species and their designated critical habitat by potentially impacting sensitive herbaceous riparian areas, it is possible that adverse impacts on sediment loading may not occur in areas where soil retention strategies are used.

6.2.8. Location of Wildlife Species

For the terrestrial exposure analysis of this risk assessment, a generic bird or mammal was assumed to occupy either the treated field or adjacent areas receiving a treatment rate on the field. Actual habitat requirements of any particular terrestrial species were not considered, and it was assumed that species occupy, exclusively and permanently, the modeled treatment area. Spray drift model predictions suggest that this assumption leads to an overestimation of exposure to species that do not occupy the treated field exclusively and permanently.

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 thiobencarb to the CRLF and DS and their designated critical habitat.

Based on the best available information, the Agency makes a **May Affect, and Likely to Adversely Affect (LAA) determination** for the CRLF and the DS from the labeled uses of thiobencarb as described in Table 7-1. The effects determination is based on potential direct and indirect effects to terrestrial-phase CRLF and potential direct and indirect effects to aquatic-phase CRLF and the DS. The LAA determination applies to all currently registered thiobencarb uses in California, *e.g.*, use of thiobencarb on rice.

Additionally, the Agency has determined that there is the potential for effects to designated critical habitat of the CRLF and the DS from the use of the thiobencarb. A summary of the risk conclusions and effects determinations for each listed species assessed and their designated critical habitat is presented in Table 7-1 and Table 7-2. Further information on the results of the effects determination is included as part of the Risk Description in Section 5.2. Given the LAA determination for the CRLF and the DS and potential designated critical habitat modification for both species, a description of the baseline status and cumulative effects for the CRLF is provided in Attachment 2 and the baseline status and cumulative effects for the DS is provided in Attachment 4.

Table 7-1. Effects Determination Summary for Potential Effects to the CRLF and DS from the Use of Thiobencarb on Rice in California

Species	Effects Determination	Basis for Determination
California red-legged frog (*Rana aurora draytonii*)	May affect, likely to adversely affect	**Potential for Direct Effects**
		Aquatic-phase (Eggs, Larvae, and Adults): Acute RQs for freshwater fish (used as a surrogate for aquatic-phase amphibians) exceed the Agency's LOCs for use of thiobencarb on rice. At the highest RQ (4.59) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.0. Chronic RQs for freshwater fish ranged from 0.26 – 46.10 and exceed the LOC of 1.0. The critical habitat and cultivated crop land cover class overlap. This indicates that direct effects to aquatic-phase CRLF have the potential to occur.
		Terrestrial-phase (Juveniles and Adults): The risk of direct adverse effects to terrestrial-phase CRLF from acute or sub-acute dietary exposure is low; however, risk may not be precluded because estimated exposure exceeds the highest doses tested where no mortality occurred for terrestrial birds (the surrogate for terrestrial-phase CRLF) consuming small insects and small mammals. The RQs for chronic risk to terrestrial birds exceed the LOC of 1.0 for birds consuming broadleaf plants/small insects and small mammals. Therefore, chronic risk to the CRLF has the potential to occur.
		Potential for Indirect Effects
		Aquatic prey items, aquatic habitat, cover and/or primary productivity Risk quotients for FW fish, FW invertebrates, and aquatic plants exceeded LOCs. For FW invertebrates, the probability of an effect would be approximately 1 in 1.0 (based on the highest RQ of 19.84 and slope of 4.5). Chronic FW invertebrate RQs also exceed the LOC of 1.0. RQs for non-vascular aquatic plants exceed the LOC of 1.0 using modeled and monitoring results. RQs for vascular aquatic plants exceed the LOC of 1.0 based on modeling data in the rice paddy. This indicates that indirect effects to CRLF have the potential to occur due to loss of prey or habitat. RQs for terrestrial plants exceed the LOC of 1.0, indicating that effects to riparian vegetation have the potential to occur.
		Terrestrial prey items, riparian habitat CRLFs could be affected as a result of potential impacts to grassy/herbaceous

Species	Effects Determination	Basis for Determination
		vegetation and reduction of prey items such as small mammals or terrestrial invertebrates. RQs for mammals consuming short grass, tall grass, broadleaf plants, and small insects exceed the acute LOC of 0.1 and chronic LOC of 1.0. The probability of individual effects for mammals is 1 in 2.73 (based on the highest RQ of 0.40 and slope of 4.5). The risk of indirect effects to the CRLF due to a reduction in terrestrial invertebrate prey items is low. Risk may not be precluded for terrestrial invertebrates because the ratio of the EEC to the dose where 15% mortality occurred exceeds the LOC of 0.05. Fifteen percent mortality occurred at the highest dose tested. It is uncertain whether the EC_{50} would result in LOC exceedances for terrestrial invertebrates. RQs for terrestrial plants exceed the corresponding LOC of 1.0.
Delta Smelt (*Hypomesus transpacificus*)	May affect, likely to adversely affect	**Potential for Direct Effects**
		RQs for freshwater and E/M fish exceed the Agency's LOCs for use of thiobencarb on rice. At the highest RQ (2.82) and using the default slope (4.5), the probability of an effect would be approximately 1 in 1.02. Chronic RQs for freshwater and estuarine/marine fish ranged from 0.26 – 13.33 and exceed the LOC of 1.0 when based on aquatic dissipation studies. Critical habitat and the cultivated crop NLCD land cover class overlap. This indicates that direct effects to DS have the potential to occur with the use of thiobencarb on rice.
		Potential for Indirect Effects
		Use of thiobencarb on rice has the potential to adversely affect the DS by reducing available food (aquatic plants and FW and E/M invertebrates), by impacting the riparian habitat of grassy and herbaceous riparian areas, and/or by impacting water quality via effects to aquatic vegetation. Acute and chronic RQs for FW and E/M invertebrates exceed corresponding LOCs indicating that reduction in prey items may occur. For FW invertebrates, the probability of an effect would be approximately 1 in 1.0 (based on the highest RQ of 19.84 and slope of 4.5). For E/M invertebrates the probability of an individual effect is approximately 1 in 1.00 (based on the highest RQ of 3.84 and a slope of 4.5). Chronic RQs for both E/M and FW invertebrates exceed the LOC of 1.0. Some RQs for aquatic plants exceed the LOC of 1.0 indicating that effects on DS habitat and reduction in food may occur. RQs for terrestrial plants exceed the LOC of 1.0 indicating that effects to riparian vegetation have the potential to occur.

Abbreviations: FW = freshwater, E/M = estuarine/marine, CRLF = California Red Legged Frog, DS=delta smelt, RQ=risk quotient

Table 7-2. Effects Determination Summary for Thiobencarb Use and CRLF and DS Critical Habitat Impact Analysis.

Assessment Endpoint	Effects Determination	Basis for Determination
Modification of aquatic-phase PCEs (DS and CRLF)	**Habitat Modification**	As described in Table 1-1, the effects determination for the potential for thiobencarb to affect aquatic-phase CRLFs and the DS is LAA. These determinations are based on the potential for thiobencarb to indirectly affect the DS and aquatic-phase CRLF. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF and DS. Therefore, potential effects to aquatic plants and terrestrial (riparian) plants identified in this assessment could result in aquatic habitat modification.
Modification of terrestrial-phase PCE (CRLF)		As described in Table 1-1, the effects determination for the potential for thiobencarb to affect terrestrial-phase CRLFs is LAA. This determination is based on the potential for thiobencarb to directly affect terrestrial-phase CRLFs and their food supply and habitat. Additionally, the potential areas of effect overlap with critical habitat designated for the CRLF. Therefore, these potential effects could result in modification of critical habitat.

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 for the CRLF and DS. 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.

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 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 CRLF and DS 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

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9. MRID List

71-1 Avian Single Dose Oral Toxicity

MRID	Citation Reference
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57223	Fletcher, D. (1972) Report to International Minerals & Chemical Corporation: Acute Oral Toxicity Study with IMC 3950 in Mallard Ducks: IBT No. J896. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095494-G)
80847	Fletcher, D. (1972) Report to International Minerals & Chemical Corporation: Acute Oral Toxicity Study with IMC 3950 in Mallard Ducks: IBT No. J896. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095106-C)
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124720	Fletcher, D. (1972) Report to International Minerals & Chemical Corporation: Acute Oral Toxicity Study with IMC 3950 in Mallard Ducks: IBT No. J896. (Unpublished study received Mar 18, 1976 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, CA; CDL: 095086-D)
42600201	Campbell, S.; Jaber, M. (1992) Bolero Technical: An Acute Oral Toxicity Study with the Northern Bobwhite: Lab Project Number: 263-126. Unpublished study prepared by Wildlife International Ltd. 18 p.
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71-2 Avian Dietary Toxicity

MRID	Citation Reference
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25775	Beavers, J.B.; Fink, R.; Grimes, J.; et al. (1979) Final Report: Subacute Feeding--Reproduction Screening Bioassay--Bobwhite Quail: Project No. 162-111. Rev. (Unpublished study including letters dated Feb 14, 1979 from F.X. Kamienski to Bob Fink and dated Mar 23, 1979 from C.A. Rohde to Joann B. Beavers, received Dec 11, 1979 under 239-2450; prepared by Wildlife International, Ltd., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 241489-D)
25779	Fletcher, D. (1976) Report to Chevron Chemical Company: 8-Day Dietary LCI50^ Study with Bolero 8 EC:STAM F-34 in Mallard Duck- lings: IBT No. 8580-09342. (Unpublished study received Dec 11, 1979 under 239-2450; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241489-J)
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57224	Fletcher, D. (1974) Report to Chevron Chemical Company, Ortho Division: 8-Day Dietary LCI50^ Study with Bolero in Bobwhite Quail: IBT No. 651-05214. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095494-H)
57225	Fletcher, D. (1974) Report to Chevron Chemical Company, Ortho Division: 8-Day Dietary LCI50^ Study with Bolero in Mallard Duck- lings: IBT No. 651-05213. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Corp., Richmond, Calif.; CDL:095494-I)

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71-4 Avian Reproduction

MRID Citation Reference

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43075401	Bavers, J.; Chafey, K.; Mitchell, L.; et al. (1993) A Reproduction Study With The Northern Bobwhite: Lab Project Number: 263-128. Unpublished study prepared by Wildlife International Ltd. 188 p.
45140601	Helsten, B. (2000) Thiobencarb: An Avian Reproductive Toxicity Study in Mallards: Lab Project Number: 133-005-08: 200000241: VP-12123. Unpublished study prepared by Bio-Life Associates, Ltd. 567 p. {OPPTS 850.2300}
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72-1 Acute Toxicity to Freshwater Fish

MRID	Citation Reference
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80852	Hamlin, J. (1971) Report to International Minerals & Chemical Corporation: Four-day Static Fish Toxicity Studies with IMC-3950 8EC in Channel Catfish and Bluegills: IBT No. A830. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095106-M)
80859	Watari, N.; Shinohara, R.; Kojima, K. (1974) Fish Toxicity Studies on Technical Product, Their By-Products and Some Potential Metabolites of Benthiocarb in the Carp and Bluegill: Toxicological Study Part II. (Unpublished paper presented at the annual meeting of the Japanese Society of Toxicology; Feb 5, 1974, Tokyo, Japan; unpublished study received Mar 18, 1976 under 239-2449; prepared by Kumiai Chemical Industry Co., Ltd., Japan, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095106-T)
116143	Hamlin, J. (1971) Report to ...: Four-day Static Fish Toxicity Studies with IMC 3950 8EC in Channel Catfish and Bluegills: IBT No. A830. (Unpublished study received Aug 3, 1972 under 2G1231; prepared by Industrial Bio-Test Laboratories, Inc., submitted by International Minerals & Chemical Corp., Libertyville, IL; CDL: 091083-G)
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155428	Chevron Chemical Co. (1986) Bolero 8EC: Wildlife & Aquatic Organ- isms Data. Unpublished compilation. 680 p.
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72-2 Acute Toxicity to Freshwater Invertebrates

MRID	Citation Reference
79118	Wheeler, R.E. (1978) 48 Hour Acute Static Toxicity of Bolero 8EC (SX981) to 1st Stage Nymph Water Fleas (Daphnia magna). (Unpublished study received Dec 11, 1979 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 241494-V)
80851	Johnson, W.W. (1973) Letter sent to H.T. Huang dated Apr 9, 1973 ?Toxicity data of IMC-3950 and Bolero 8EC to freshwater fish and crayfish|. (U.S. Fish and Wildlife Service, Fish-Pesticide Re- search Laboratory; unpublished study; CDL:095106-L)
80858	Ward, S. (1975) Acute Toxicity of Bolero 8-emulsive to Four Species of Decapod Crustaceans. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Bionomics--EG & G, Inc., sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:095106-S)
85633	Wheeler, R.E. (1978) 48 Hour Acute Static Toxicity of Bolero (SX796) to 1st Stage Nymph Water Fleas (*Daphnia magna*). (Unpublished study received Dec 11, 1979 under 239-2449; submit- ted by Chevron Chemical Co., Richmond, Calif.; CDL:241494-U)
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72-3 Acute Toxicity to Estuarine/Marine Organisms

MRID	Citation Reference
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79113	Heitmuller, T. (1979) Acute Toxicity of Bolero 8 EC to Fiddler Crabs (Uca pugilator): Report No. BP-79-9-135. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by EG & G, Bionomics, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241494-P)
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79116	Lauck, J.E. (1979) Field Bioassay-palaeomonid Grass Shrimp. (Un- published study received Dec 11, 1979 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241494-S)

80853 Rausina, G. (1974) Report to Chevron Chemical Company, Ortho Division: Four-day Static Aquatic Toxicity Study with XE-362 Technical (Benthiocarb) in Eastern Oysters: IBT No. 621-05226. (Un- published study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095106-N)

80854 Rausina, G. (1974) Report to Chevron Chemical Company, Ortho Division: Four-day Static Aquatic Toxicity Study with Benthiocarb Technical in Shore Crabs: IBT No. 621-05226. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095106-O)

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40651314 Bailey, H. (1988) Acute Toxicity of Rice Herbicides to Neomysis mercedis: Laboratory Project ID: R and RA 88-14. Unpublished study prepared by SRI International. 28 p.

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81-1 Acute oral toxicity in rats

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81-3 Acute inhalation toxicity in rats

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92182027	Kodama, J. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00040585 and Related MRIDs 00134976. The Acute Inhalation Toxicity of BOLERO Technical Vapor: Project No. SOCAL 885. Prepared by Chevron Environmental Health Center, Inc. 9 p.
92182028	Kodama, J. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00161698. The Acute Inhalation Toxicity of BOLERO 8EC (PN 5281) in Rats: Project No. SOCAL 1960. Prepared by Chevron Environmental Health Center, Inc. 13 p.

Neurotoxicity study in hens

MRID	Citation Reference
84135	Ben-Dyke, R.; Bagley, K.; Cavanagh, J.B. (1978) Bolero: Examination for Potential To Cause Delayed Neurotoxicity in Hens: LSR Report No. 78/KCI26/407. (Unpublished study received Dec 11, 1979 under 6F1763; prepared by Life Science Research, England, sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:099125-K)
92182060	Kodama, J. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00135284 and Related MRIDs 00084135. BOLERO: Examination for Potential to Cause Delayed Neurotoxicity in Hens: Project No. 78/KCI 26/407. Prepared by Life Science Research. 12 p.
92182076	Kodama, J. (1990) Chevron Chemical Company Phase 3 Reformat of MRID 00135284 and Related MRIDs 00084135. BOLERO: Examination for Potential to Cause Delayed Neurotoxicity in Hens: Project No. 78/KCI 26/407. Prepared by Life Science Research. 30 p.

81-8 Acute neurotoxicity screen study in rats

MRID	Citation Reference
42987001	Lamb, I. (1993) An Acute Neurotoxicity Study of BOLERO Technical in Rats: Lab Project Number: WIL-194010: VP-10007. Unpublished study prepared by WIL Research Labs, Inc. 1052 p.
43148202	Lamb, I. (1994) A Range-Finding Study of BOLERO Technical in Rats: Lab Project Number: WIL/194009. Unpublished study prepared by WIL Research Lab., Inc. 206 p.

82-1 Subchronic Oral Toxicity: 90-Day Study

MRID	Citation Reference
40588	Smith, P.S.; Yost, D.H. (1972) Report to International Minerals & Chemical Corporation: 90-Day Subacute Oral Toxicity Study with IMC 3950 Technical in Albino Rats: IBT No. B353. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Indus- trial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095493-W)
40589	Hartke, K.; Gordon, D.E. (1972) Report to International Minerals & Chemical Corporation: 90-Day Subacute Oral Toxicity Study with IMC 3950 Technical in Beagle Dogs: IBT No. C610. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Indus- trial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond Calif.; CDL:095493-Y)
41176	McCollum, K. (1973) Report to International Minerals & Chemical Corporation: 21-Day Paired-Feeding Study with IMC 3950 in Albino Rats: IBT No. 621-03628. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095491-F)
41178	Kieckebusch, W.; Griem, W.; Lang, K. (1975) The Toxicity of p-Chlo- robenzoic acid. A translation of: Die Vertraglichkeit der p-Chlorbenzoesaure. Arzneitmittel Forschung 10(12):999-1001. (Unpublished study including German text, received Mar 18, 1976 under 239-EX-78; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095491-H)
69694	McCollum, K. (1973) Report to International Minerals & Chemical Corporation: 21-day Paired-feeding Study with IMC 3950 in Al- bino Rats: IBT No. 621-03628. (Unpublished study received Mar 18, 1976 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095085-F)
84130	Cummins, H.A.; Ashby, R. (1979) Bolero^(R)I: Toxicity in Dietary Administration to Rats for Up to Eight Weeks (Range-finding Study): LSR Report No. 79/KC127/074. (Unpublished study received Dec 11, 1979 under 6F1763; prepared by Life Science Research, England, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099125-F)
86870	Cummins, H.A.; Ashby, R. (1980) Bolero^(R)I: Combined Oncogenicity and Chronic Feeding Study in the Rat: Summary Report after Premature Termination after 25 Weeks of Treatment: LSR Report No. 80/KCI028/207. (Unpublished study received Nov 30, 1981 under 0F2322; prepared by Life Science Research, England, sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:070486-A)
86871	Cummins, H.A.; Ashby, R.; Finn, J.P.; et al. (1980) Bolero^(R)I: Palatability Study by Paired Feeding in the Rat: LSR Report No. 80/KCI043/034. Final rept. (Unpublished study received Nov 30, 1981 under 0F2322; prepared by Life Science Research, England, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:070487-A)

MRID	Citation Reference
116141	Industrial Bio-Test Laboratories, Inc. (1971) Study Review: IMC 3950|: IBT No. B-353. (Unpublished study received Aug 3, 1972 under 2G1231; submitted by International Minerals & Chemical Corp., Libertyville, IL; CDL:091083-E)
116142	Industrial Bio-Test Laboratories, Inc. (1971) Study Review: IMC 3950|: IBT No. C-610. (Unpublished study received Aug 3, 1972 under 2G1231; submitted by International Minerals & Chemical Corp., Libertyville, IL; CDL:091083-F)
116150	Smith, P. (1972) Report to ...: 90-day Subacute Oral Toxicity Study with IMC 3950 Technical in Albino Rats: IBT No. B353. (Unpublished study received Aug 5, 1972 under 2G1231; prepared by Industrial Bio-Test Laboratories, Inc., submitted by International Minerals & Chemical Corp., Libertyville, IL; CDL:091085-H)
116151	Hartke, K. (1972) Report to ...: 90-day Subacute Oral Toxicity Study with IMC 3950 Technical in Beagle Dogs: IBT No. C610. (Unpublished study received Aug 5, 1972 under 2G1231; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Inter- national Minerals and Chemical Corp., Libertyville, IL; CDL: 091085-I)
144742	Johnson, D. (1985) One Year Subchronic Oral Toxicity Study with Thiobencarb Technical in Dogs: 415-042. Unpublished study pre- pared by International Research and Development Corp. 408 p.
164576	Richter, W. (1972) 90-Day Subacute Oral Toxicity Study with IMC 3950 Technical in Albino Rats: Addendum Report to International Minerals & Chemical Corp.: P.O. No. ILV-001305: IBT No. B353. Unpublished study prepared by Industrial Bio-Test Laboratories, Inc. 4 p.

82-2 21-day dermal-rabbit/rat

MRID	Citation Reference
40590	Lukes, T.H.; Paa, H.; Robl, M.G. (1974) Report to Chevron Chemical Company: 21-Day Subacute Dermal Toxicity Study with Bolero 8 Emulsive (Benthiocarb 8 Emulsive, XE-362 8 Emulsive) in Albino Rabbits: IBT No. 601-05223. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Industrial Bio-Test Indus- tries, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095493-Z)
134977	Lukes, T.; Paa, H. (1974) Report to ...: 21-Day Subacute Dermal Toxicity Study with Bolero & Emulsive (Benthiocarb & Emulsive, XE-362 & Emulsive) in Albino Rabbits: IBT No. 601-05223. (Un- published study received Mar 18, 1976 under 239-EX-77; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, CA; CDL:095492-Y)
42003401	Machado, M. (1991) Three-Week Repeated-Dose Dermal Toxicity Study in Adult Male and Female Rats with Bolero 8EC (SX-1843): Lab Project Number: CEHC/3142. Unpublished study prepared by Chevron Environmental Health Center, Inc. 382 p.
42893001	Machado, M. (1993) Three-Week Repeated-Dose Dermal Toxicity Study in Adult Male and Female Rats with BOLERO 8EC (SX-1843) MRID 42003401: Revised Report Number One: Lab Project Number: 5510. Unpublished study prepared by Chevron Research & Technology Co. 391 p.

82-4 90-day inhal.-rat

MRID	Citation Reference

40591	Churukian, P.V.; Arceo, R. (1974) Report to Chevron Chemical Company, Ortho Division: 14-Day Subacute Aerosol Inhalation Toxicity Study with Bolero 8 EC (Benthiocarb 8EC, XE-362 8 EC) in Albino Rats: IBT No. 663-05224. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095493-AA)

82-5 Subchronic Neurotoxicity: 90-Day Study

MRID	Citation Reference
84134	Fletcher, D.; Arceo, R.J. (1977) Report to Kumiai Chemical Indus- tries Company, Inc.: Neurotoxicity Study with Bolero Technical in Chickens: IBT No. 8580-10025. (Unpublished study received Dec 11, 1979 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099125-J)
135283	Fletcher, D.; Arceo, R. (1977) Report to Kumiai Chemical Industries Company, Inc.: Neurotoxicity Study with Bolero Technical in Chickens: IBT No. 8580-10025. (Unpublished study received Dec 1, 1978 under 239-EX-77; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, CA; CDL:097658-C)

82-7 Subchronic Neurotoxicity

MRID	Citation Reference
43001001	Lamb, I. (1993) A Subchronic (13-Week) Neurotoxicity Study of Bolero Technical in Rats: Lab Project Number: WIL/194011: 194011: VP/10008. Unpublished study prepared by WIL Research Labs., Inc. 1634 p.

83-1 Chronic Toxicity

MRID	Citation Reference
41175	Morrow, L.; Arceo, R.J. (1974) Report to Chevron Chemical Company, Ortho Division: Two-Year Chronic Oral Toxicity Study with Benthiocarb Technical (XE-362 Technical, Bolero Technical) in Albi- no Rats: IBT No. 621-02095. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095491-E)
41177	Mastalski, K.; Robl, M.G. (1974) Report to Chevron Chemical Company, Ortho Division: Two-Year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb) in Beagle Dogs: IBT No. 651-02096. (Unpublished study received Mar 18, 1976 under 239-EX-78; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095491-G)
69693	Morrow, L. (1974) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with Benthiocarb Technical (XE-362 Technical, Bolero Technical) in Albino Rats: IBT No. 621-02095. (Unpublished study received Mar 18, 1976 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095085-E)
72675	Morrow, L.; Sullivan, D.J. (1976) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with Bolero (XE-362, Benthiocarb) in Albino Rats: IBT No. 621-04652. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Industrial

Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 241496-A)

72676 Mastalski, K.; Richter, W.R. (1976) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb in Beagle Dogs): IBT No. 651-05143. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 241496-B)

81902 Mastalski, K.; Robl, M.G. (1974) Report To Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb) in Beagle Dogs: IBT No. 651-02096. (Unpublished study received Jan 10, 1975 under 5G1582; prepared by Industrial Bio-Test Laboratories, Inc., sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:095344-E)

82633 Mastalski, K. (1974) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb) in Beagle Dogs: IBT No. 651-02096. (Unpublished study received Mar 18, 1976 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095085-G)

83625 Morrow, L. (1976) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity with Bolero (XE-362, Benthiocarb) in Albino Rats: IBT No. 621-04652. (Unpublished study received Dec 12, 1979 under 6F1763; prepared by Industrial Bio- Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099126-A)

83626 Mastalski, K. (1976) Report to Chevron Chemical Company, Ortho Division: Two-year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb) in Beagle Dogs: IBT No. 651- 05143. (Unpublished study received Dec 12, 1979 under 6F1763; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099126-B)

86004 Macrae, S.M.; Amyes, S.J.; Holmes, P.; et al. (1981) Technical Bolero(R): Potential Oncogenicity in Dietary Administration to Mice: LSR Report No. 81/KCI040/527. Final rept. (Unpublished study received Nov 30, 1981 under 0F2322; prepared by Life Science Research, England, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:070480-A; 070479; 070483; 070484; 070485; 070491; 070488; 070489)

86821 Cummins, H.A.; Bhatt, A.; Afzaal, M.; et al. (1981) Technical Bolero^(R)I: Combined Oncogenicity and Toxicity Study in Dietary Ad- ministration to the Rat: 81/KCI045/478. Interim rept. 3: 0-52 weeks. (Unpublished study received Nov 30, 1981 under 0F2322; prepared by Life Science Research, England, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:070493-A; 070482; 070481)

108670 Morrow, L.; Sullivan, D. (1976) Report to ..., Ortho Division: Two-year Chronic Oral Toxicity Study with Bolero (XE-362, Benthiocarb) in Albino Rats: IBT No. 621-04652. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, CA; CDL:241486-A)

108671 Mastalski, K.; Richter, W. (1976) Report to ..., Ortho Division: Two-year Chronic Oral Toxicity Study with XE-362 Technical (Bolero, Benthiocarb) in Beagle Dogs: IBT No. 651-05143. (Unpublished study received Dec 11, 1979 under 239-2449; prepared by Industrial Bio-Test Laboratories, Inc., submitted by Chevron Chemical Co., Richmond, CA; CDL:241486-B)

148570 Life Science Research Ltd. (1984) Technical Bolero: Combined Oncogenicity and Toxicity Study in Dietary Administration to the Rat: Final Report: 83/KC1045/248. Unpublished study. 1094 p.

150139 Cummings, H. (1984) Combined Oncogenicity and Toxicity Study in Dietary Administration to

	the Rat: Technical Bolero: Final Report: Report No. 83/KCI045/248. Unpublished study prepared by Life Science Research Ltd. 264 p.
150894	Life Science Research Ltd. (1984) Technical Bolero: Combined Oncogenicity and Toxicity Study in Dietary Administration to the Rat Final Report: [Appendices Vol. 2 and 3]. Unpublished study. 765 p.
154506	Cummins, H. (1984) Technical Bolero: Combined Oncogenicity and Toxicity Study in Dietary Administration to the Rat: Amended Final Report: 84/KCI045/579. Unpublished study prepared by Life Science Research Ltd. 2135 p.
92182035	Kodama, J. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00154506 and Related MRIDs 00084130, 00086821, 00086870, 00086871, 00148570, 00150139, 00150894. BOLERO Technical: Combined Oncogenicity and Toxicity Study in Dietary Administration to the Rat: Amended Final Report: LSR Report No. 84/KCI 045/579. Prepared by Life Science Research. 46 p.
92182036	Kodama, J. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00144742. One-Year Subchronic Oral Toxicity Study with Thiobencarb Technical in Dogs: IRDC Study No. 415-042 (415-041: Pilot Study). Prepared by International Research and Devl. Corp. 19 p.

1 Phytotoxicity

MRID	Citation Reference
2669	Fischer, B.B. (1973) Tomato Weed Control Trial: 1973 Vegetation Management in Tomato Production: Report No. 40483. (Unpublished study received May 6, 1976 under 3125-277; prepared by (Univ. of California--Riverside), West Side Field Station, Farm Advisor Office, submitted by Mobay Chemical Corp., Agricultural Chemicals Div., Kansas City, Mo.; CDL:224187-AY)
3252	Talbert, R.E.; Kennedy, J.M. (1974) Field Evaluation of Herbicides in Vegetable Crops, 1973. By Univ. of Arkansas, Depts. of Agronomy, Horticulture and Forestry. Fayetteville, Ark.: Univ. of Arkansas, Agricultural Experiment Station. (Mimeograph series 219; also In unpublished submission received May 6, 1976 under 3125-277; submitted by Mobay Chemical Corp., Agricultural Chemicals Div., Kansas City, Mo.; CDL:224187-BD)
3463	Palmer, R.D.; Helpert, C.W. (1973) Rice Weed Control in the Western Belt of Texas. (Unpublished study received May 7, 1974 under 4G1505; prepared by Texas ?A & M Univ.|, Agricultural Experiment Station and Agricultural Extension Service, submitted by Mobil Chemical Co., Industrial Chemicals, Richmond, Va.; CDL: 093987-AG)
4176	Eastin, E.F.; Stansel, J.W.; Flinchum, W.T.; Helpert, C.W.; Young, J. (1973) Herbicide Field Evaluations for Rice, 1973: Report No.7. (Unpublished study received May 7, 1974 under 4G1505; prepared by Texas A & M Univ., Agricultural Experiment Station, Agricultural Research & Extension Center at Beaumont, submitted by Mobil Chemical Co., Industrial Chemicals, Richmond, Va.; CDL:093987-AE)
19202	Buchanan, G.A. (1971) Preplant and Preemergence Weed Control in Cotton: Test No. Lib. No. 6223. (Unpublished study received Apr 29, 1976 under 100-523; prepared by Auburn Univ., submitted by Ciba-Geigy Corp., Greensboro, N.C.; CDL:225271-V)
25179	Lauck, J.E. (1979) Final Report of Field Study: Ortho Bolero 8 EC-- Rice, 1979. (Unpublished study received Dec 11, 1979 under 239- 2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-F)
26727	Gerhold, J.F.; Gruelach, L.; Cates, M.D.; et al. (1974) Devrinol 2-E Use on Tomatoes|.

	(Unpublished study received Sep 7, 1976 under 476-2150; prepared in cooperation with Univ. of Southwestern Louisiana, submitted by Stauffer Chemical Co., Richmond, Calif.; CDL:225549-D)
26910	Buchanan, G.A. (1971) Preplant and Preemergence Weed Control in Cotton: Report No. 6223. (Unpublished study received Apr 29, 1976 under 100-523; prepared by Auburn Univ., submitted by Ciba- Geigy Corp., Greensboro, N.C.; CDL:224757-X)
31750	Locascio, S.; Layton, J.; Sheets, W.A.; et al. (1976) ?Weed Control in Strawberries with Devrinol 50 WP|. (Unpublished study received Sep 7, 1976 under 476-2108; prepared in cooperation with Oregon State Univ., North Willamette Experiment Station, submit- ted by Stauffer Chemical Co., Richmond, Calif.; CDL:225548-B)
31754	Gerhold, J.F.; Stevenson, V.C.; Greulach, L.; et al. (1974) ?Weed Control in Transplanted Tomatoes with Devrinol + Tillam Tank Mix|. (Unpublished study received Sep 7, 1976 under 476-2108; prepared in cooperation with Univ. of Missouri, submitted by Stauffer Chemical Co., Richmond, Calif.; CDL:225548-F)
31761	Gerhold, J.; Monaco, T.; Hickey, S.; et al. (1974) Devrinol 2E Pre- plant Incorporated on Tomatoes/Peppers. (Unpublished study received Sep 7, 1976 under 476-2108; submitted by Stauffer Chemical Co., Richmond, Calif.; CDL:225348-M)
124051	Chevron Chemical Co. (1976) ?Bolero 10G: Phytotoxicity|. (Compilation; unpublished study received Mar 18, 1976 under 239-2449; CDL:095105-A)
124279	Chevron Chemical Co. (1982) Efficacy Data Reports: Bolero 8 EC/ Rice. (Compilation; unpublished study received Jan 24, 1983 under 239-EX-77; CDL:249348-B)
124597	Chevron Chemical Co. (1974) Bolero Herbicide: Amount, Frequency and Timing of Application of the Pesticide Chemical. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094341-A)
134990	Chevron Chemical Co. (1976) Phytotoxicity Data: Weed Control in Dry-seeded Rice in California ?Using Bolero 10 G|. (Compilation; unpublished study received Mar 18, 1976 under 239-2449; CDL:095724-A)
134991	Chevron Chemical Co. (1976) Efficacy Data of Bolero & EC for Weed Control on Rice Including a Review of the Literature under Benthiocarb|. (Compilation; unpublished study received Mar 18, 1976 under 239-2450; CDL:095725-A)
134992	Chevron Chemical Co. (1976) Bolero & EC: Efficacy Data: Rice|. (Compilation; unpublished study received Mar 18, 1976 under 239- 2450; CDL:095726-A)
134993	Chevron Chemical Co. (1976) Bolero 10 G: Efficacy Data: Rice|. (Compilation; unpublished study received Mar 18, 1976 under 239- 2449; CDL:095727-A)
138506	International Minerals & Chemical Corp. (1971) Use of Bolero & EC on Rice Fields for Control of Weeds. (Compilation; unpublished study received Aug 3, 1972 under 2G1231; CDL:091082-A)

122-1 Seed Germination/Seedling Emergence and Vegetable Vigor

MRID	Citation Reference
41690902	Hoberg, J. (1990) Thiobencarb Technical: Determination of Effects on Seed Germination, Seedling Emergence and Vegetative Vigor of Ten Plant Species: Lab Project Number: 90-9-3462. Unpublished study prepared by Springborn

Laboratories, Inc. 170 p.

122-2 Aquatic plant growth

MRID	Citation Reference
41690901	Giddings, J. (1990) Thiobencarb Technical-Toxicity to Five Species of Aquatic Plants: Lab Project Number: 90-9-3477. Unpublished study prepared by Springborn Laboratories, Inc. 116 p.
41690902	Hoberg, J. (1990) Thiobencarb Technical: Determination of Effects on Seed Germination, Seedling Emergence and Vegetative Vigor of Ten Plant Species: Lab Project Number: 90-9-3462. Unpublished study prepared by Springborn Laboratories, Inc. 170 p.

123-1 Seed germination/seedling emergence and vegetative vigor

MRID	Citation Reference
41690902	Hoberg, J. (1990) Thiobencarb Technical: Determination of Effects on Seed Germination, Seedling Emergence and Vegetative Vigor of Ten Plant Species: Lab Project Number: 90-9-3462. Unpublished study prepared by Springborn Laboratories, Inc. 170 p.
44846201	Chetram, R. (1999) Tier 2 Seedling Emergence Nontarget Phytotoxicity Study Using Thiobencarb: Lab Project Number: 98743: 9900179. Unpublished study prepared by ABC Laboratories. 79 p.

123-2 Aquatic plant growth

MRID	Citation Reference
41690901	Giddings, J. (1990) Thiobencarb Technical-Toxicity to Five Species of Aquatic Plants: Lab Project Number: 90-9-3477. Unpublished study prepared by Springborn Laboratories, Inc. 116 p.
41690902	Hoberg, J. (1990) Thiobencarb Technical: Determination of Effects on Seed Germination, Seedling Emergence and Vegetative Vigor of Ten Plant Species: Lab Project Number: 90-9-3462. Unpublished study prepared by Springborn Laboratories, Inc. 170 p.

161-1 Hydrolysis

MRID	Citation Reference
39098	Pack, D.E. (1974) The Stability of Benthiocarb in Water. Includes method entitled: Determination of Benthiocarb in water. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-A)
39099	Cheng, H.M. (1976) Photodegradation Studies with ?Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239- 2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095088-B)
39100	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and

	Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2450; prepared by Kumiai Chemical Industry Co., Ltd. and Nagoya Univ., Faculty of Agriculture, Laboratory of Soil Science, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-C)
40923	Cheng, H.M. (1976) Photodegradation Studies with Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-B)
44493	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Nagoya Univ., Laboratory of Soil Science and Kumiai Chemical Industry Co., Ltd., submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095096-L)
44494	Cheng, H.M. (1976) Photodegradation Studies with Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239- 2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095096-M)
44498	Pack, D.E. (1974) The Stability of Benthiocarb in Water: File No. 741.10. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095095-B)
87205	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1977) Photodegradation of benthiocarb herbicide. Journal of Pesticide Science 2 (1):17-25. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-K)
87206	Ishikawa, K.; Nakamura, Y.; Kuwatsuka, S. (1977) Volatilization of benthiocarb herbicide from the aqueous solution and soil. Journal of Pesticide Science 2(2):127-134. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-L)
123979	Chevron Chemical Co. (1974) Bolero Herbicide: The Results of Tests on the Amount of Residue Remaining, Including a Description of the Analytical Methods Used. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094345-A; 094342)
124724	Pack, D. (1974) The Stability of Benthiocarb in Water: File No. 741.10. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, CA; CDL:095102-B)
124730	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2450; prepared by Kumiai Chemical Industry Co., Ltd., Japan, submitted by Chevron Chemical Co., Richmond, CA; CDL:095103-G)
125180	Chevron Chemical Co. (1974) Bolero Herbicide: Name, Chemical Identity and Composition of the Pesticide Chemical. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL: 094339-B)
152311	Crosby, D. (1983) The fate of herbicides in California rice culture. P. 339-346 in Pestic. Chem.: Human Welfare Environ. Proc. Int. Cong. Pestic. Chem., 5th 1982, v. 2, ed. by J. Miyamoto; P. Kearney. Pergamon: Oxford, UK.
41609012	Chen, Y. (1990) Hydrolysis of ?Phenyl-U-?carbon 14||-Thiobencarb in Water: Lab Project Number: MEF-0149/9007557. Unpublished study prepared by Chevron Chemical Co. 32 p.

161-2 Photodegradation-water

MRID	Citation Reference

123979	Chevron Chemical Co. (1974) Bolero Herbicide: The Results of Tests on the Amount of Residue Remaining, Including a Description of the Analytical Methods Used. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094345-A; 094342)
152311	Crosby, D. (1983) The fate of herbicides in California rice culture. P. 339-346 in Pestic. Chem.: Human Welfare Environ. Proc. Int. Cong. Pestic. Chem., 5th 1982, v. 2, ed. by J. Miyamoto; P. Kearney. Pergamon: Oxford, UK.
42257801	Chen, Y. (1988) Photodegradation of Phenyl-U-14C|-Thiobencarb in Water: Lab Project Number: MEF-0091. Unpublished study prepared by Chevron Chemical Co. 45 p.

161-3 Photodegradation-soil

MRID	Citation Reference
39099	Cheng, H.M. (1976) Photodegradation Studies with ?Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239- 2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095088-B)
40923	Cheng, H.M. (1976) Photodegradation Studies with ?Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-B)
40924	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Kumiai Chemical Industry Co., Ltd. and Nagoya Univ., Laboratory of Soil Science, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-C)
44494	Cheng, H.M. (1976) Photodegradation Studies with ?Ring-U-14C| Benthiocarb. (Unpublished study received Mar 18, 1976 under 239- 2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095096-M)
87204	Nakamura, Y.; Ishikawa, K.; Kuwatsuku, S. (1977) Degradation of benthiocarb in soils as affected by soil conditions. Journal of Pesticide Science 2(1):7-16. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-J)
87205	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1977) Photodegradation of benthiocarb herbicide. Journal of Pesticide Science 2 (1):17-25. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-K)
87206	Ishikawa, K.; Nakamura, Y.; Kuwatsuka, S. (1977) Volatilization of benthiocarb herbicide from the aqueous solution and soil. Journal of Pesticide Science 2(2):127-134. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-L)
124730	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2450; prepared by Kumiai Chemical Industry Co., Ltd., Japan, submitted by Chevron Chemical Co., Richmond, CA; CDL:095103-G)
41215312	McGovern, P. (1988) Soil Surface Photolysis of ?Carbon 14|-Thiobencarb in Natural Sunlight: Project ID MEF-0010. Unpublished study prepared by the Pharmacology and Toxicology Research Laboratory. 89 p.
92182047	Chen, Y. (1990) Chevron Chemical Company Phase 3 Summary of MRID 41215312. Soil Surface Photolysis of Carbon-14 Thiobencarb in Natural Sunlight, PTRL 129W. Prepared by

Pharmacology and Toxicology Research Lab. 16 p.

162-1 Aerobic soil metabolism

MRID	Citation Reference
39101	Pack, D.E. (1975) The Soil Metabolism of ?Ring-U-14C|Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2450; submit- ted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-D)
39102	Pack, D.E. (1974) A Comparison of the Rates of Soil Degradation of Benthiocarb under flooded vs Non-Flooded and Aerobic v. Anaerobic Conditions. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-E)
39103	Pack, D.E. (1974) The Effect of Sterilization on the Rate of Soil Degradation of Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-F)
39104	Nakamura, Y.; Kuwatsuka, S. (1974) Degradation of Benthiocarb in Soil. (Unpublished study received Mar 18, 1976 under 239-2450; prepared by Nagoya Univ. and Kumiai Chemical Industry Co., Ltd., submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095088-G)
87204	Nakamura, Y.; Ishikawa, K.; Kuwatsuku, S. (1977) Degradation of benthiocarb in soils as affected by soil conditions. Journal of Pesticide Science 2(1):7-16. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-J)
96954	Pack, D.E. (1974) The Soil Metabolism of Ring-UL^14IC|-benthiocarb: File No. 773.21. (Unpublished study received Feb 14, 1975 under 239-EX-77; CDL:226080-A)
123979	Chevron Chemical Co. (1974) Bolero Herbicide: The Results of Tests on the Amount of Residue Remaining, Including a Description of the Analytical Methods Used. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094345-A; 094342)
135374	Pack, D. (1975) The Soil Metabolism of ?Ring-U-14C|Benthiocarb: File No. 773.21. (Unpublished study received Mar 18, 1976 under 6F1763; submitted by Chevron Chemical Co., Richmond, CA; CDL: 098089-D)
43300401	Patterson, T. (1994) Aerobic Metabolism of (Phenyl-(carbon 14))-Thiobencarb in Soil: Lab Project Number: PRT-08-2VNA-01: 10210: PRT-08-2VNA-01-011. Unpublished study prepared by Plant Research Technologies, Inc. 145 p.
92182048	Pack, D. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00040925. The Soil Metabolism of Ring-U-Carbon-14 Benthiocarb. Prepared by Chevron Chemical Company. 18 p.

162-3 Anaerobic aquatic metab.

MRID	Citation Reference
43252001	Esser, T.; Shepler, K. (1994) Anaerobic Aquatic Metabolism of (phenyl-carbon 14) Thiobencarb: Lab Project Number: 397W: VP-10505. Unpublished study prepared by PTRL West, Inc. 116 p.
92182049	Pack, D. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00040925. The Soil Metabolism of Ring-U-Carbon-14 Benthiocarb. Prepared by Chevron Chemical Company. 17 p.

162-4 Aerobic aquatic metab.

MRID	Citation Reference
42015301	Mulkey, N. (1991) The Aerobic Aquatic Metabolism of ?Ring-Carbon 14|-Thiobencarb: Lab Project Number: ADC 1238. Unpublished study prepared by Analytical Development Corp. 159 p.

163-1 Leach/adsorp/desorption

MRID	Citation Reference
39100	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2450; prepared by Kumiai Chemical Industry Co., Ltd. and Nagoya Univ., Faculty of Agriculture, Laboratory of Soil Science, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-C)
39106	Warnock, R.E. (1975) Adsorption, Desorption and Freundlich Constants of Benthiocarb in Soil. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-K)
40928	Warnock, R.E. (1974) Mobility of Benthiocarb and pCl-Benzoic acid in Soil As Determined by Soil TLC Techniques. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-I)
40929	Warnock, R.E. (1974) Benthiocarb Leaching Study--EPA Protocol. (Unpublished study received Mar 18, 1976 under 239-2449; sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-J)
40930	Warnock, R.E. (1975) Adsorption, Desorption and Freundlich Constants of Benthiocarb in Soil. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-K)
40931	Warnock, R.E. (1975) Volatilization of Benthiocarb--Laboratory Studies. (Unpublished study received Mar 18, 1976 under 239- 2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 095091-L)
41813	Warnock, R.E. (1974) Benthiocarb Leaching Study--EPA Protocol. (Unpublished study received Mar 18, 1976 under 239-2450; submit- ted by Chevron Chemical Co., Richmond, Calif.; CDL:095101-J)
44500	Warnock, R.E. (1975) Adsorption, Desorption and Freundlich Constants of Benthiocarb in Soil: File No. 741.11. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095095-D)
44501	Warnock, R.E. (1975) Volatilization of Benthiocarb--Laboratory Studies: File No. 721.2 (Bolero). (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095095-E)
44511	Pack, D.E. (1975) The Soil Metabolism of ?Ring-U-14C| Benthiocarb: File No. 773.21. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095094-B)
44512	Warnock, R.E. (1974) Benthiocarb Leaching Study--EPA Protocol: File No. 721.14 (Bolero). (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095094-J)

MRID	Citation Reference
86782	Sachs, E.S. (1976) Comparative Soil Sorption, Movement and Volatility of Three Thiocarbamate Herbicides. Master's thesis, Univ. of California--Davis. (Unpublished study received Oct 13, 1981 under 2G2583; submitted by Sierra Chemical Co., Milpitas, Calif.; CDL:070392-J)
87206	Ishikawa, K.; Nakamura, Y.; Kuwatsuka, S. (1977) Volatilization of benthiocarb herbicide from the aqueous solution and soil. Journal of Pesticide Science 2(2):127-134. (Also In unpublished submission received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:099129-L)
96972	Pack, D.E. (1977) Soil Mobility of Captan, Folpet and Captafol As Determined by Soil Thin-layer Chromatography: File No. 722.0. (Unpublished study received May 30, 1978 under 239-2211; sub- mitted by Chevron Chemical Co., Richmond, Calif.; CDL:234046-N)
123979	Chevron Chemical Co. (1974) Bolero Herbicide: The Results of Tests on the Amount of Residue Remaining, Including a Description of the Analytical Methods Used. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094345-A; 094342)
124725	Warnock, R. (1975) Adsorption, Desorption and Freundlich Constants of Benthiocarb in Soil: File No. 741.11. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, CA; CDL:095102-C)
124728	Warnock, R. (1975) 14C-Benthiocarb Residues in Soil and Uptake by Carrots--EPA Protocol: File No. 741.11/Bolero/Carrots. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, CA; CDL:095103-C)
135375	Warnock, R. (1974) Benthiocarb Leaching Study: File No. 721.14 (Bolero). (Unpublished study received Mar 18, 1976 under 6F1763; submitted by Chevron Chemical Co., Richmond, CA; CDL: 098089-J)
135376	Warnock, R. (1975) Volatilization of Benthiocarb--Laboratory Studies: File No. 721.2 (Bolero). (Unpublished study received Mar 18, 1976 under 6F1763; submitted by Chevron Chemical Co., Richmond, CA; CDL:098089-L)
41215313	Pack, D. (1988) Freundlich Soil Adsorption/Desorption Coefficients of Thiobencarb: Project ID MEF-0097/8814700. Unpublished study prepared by Chevron Chemical Co. 37 p.
43121201	Schocken, M. (1994) (carbon 14) Thiobencarb--Determination of Adsorption/Desorption Coefficients for Soil Degradates: Lab Project Number: VP-10011: 93-12-5074. Unpublished study prepared by Springborn Laboratories, Inc. 71 p.
43150601	Christensen, K. (1994) 4-Chlorobenzoic Acid--Determination of the Adsorption and Desorption Properties: Lab Project Number: 12709.1093.6132.710: 93-11-5051: VP-10804. Unpublished study prepared by Springborn Labs., Inc. 73 p.
92182050	Pack, D. (1990) Chevron Chemical Company Phase 3 Summary of MRID 41215313. Freundlich Adsorption/Desorption Coefficients of Thiobencarb: Project No. MEF-0097/888814700. Prepared by Chevron Chemical Co. 12 p.

163-2 Volatility - lab

MRID	Citation Reference
40924	Ishikawa, K.; Nakamura, Y.; Niki, Y.; et al. (1973) Benthiocarb Volatilization from and Photodecomposition in Aqueous Solution. Rev. (Unpublished study received Mar 18, 1976 under 239-2449; prepared by Kumiai Chemical Industry Co., Ltd. and Nagoya Univ., Laboratory of Soil Science, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-C)

123979	Chevron Chemical Co. (1974) Bolero Herbicide: The Results of Tests on the Amount of Residue Remaining, Including a Description of the Analytical Methods Used. (Compilation; unpublished study received Jan 10, 1975 under 5G1582; CDL:094345-A; 094342)

164-1 Terrestrial field dissipation

MRID	Citation Reference
25182	Serat, W.F.; Howard, J.; Cooper, W.D.; et al. (1975) Experimental Use Permit Health Effects: ?Bolero|. (Unpublished study received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-I)
32808	Chevron Chemical Company (19??) Recovery of Bolero 10G and Bolero 8EC from Rice Field: Substrata. (Unpublished study received Dec 11, 1979 under 239-2450; CDL:241498-C)
39105	Tucker, B.V. (1974) Simulated Field Study of Rice Treated with 14C- Ring-Labeled Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095088-H)
40927	Tucker, B.V. (1974) Simulated Field Study of Rice Treated with 14C-Ring-Labeled Benthiocarb. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095091-H)
72530	Serat, W.F. (1975) Work Unit D--Experimental Use Permit Health Effects: ?Bolero in Rice|. (Unpublished study received Dec 11, 1979 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241499-B)
79100	Chevron Chemical Company (1975) Work Unit D--Experimental Use Permit Health Effects. (Compilation; unpublished study received Dec 11, 1979 under 239-2449; CDL:241495-G)
116146	International Minerals & Chemical Corp. (1971) ?Study: p-Chlorobenzyl Alcohol Residues in Rice|. (Compilation; unpublished study received Aug 5, 1972 under 2G1231; CDL:091085-D)
124233	Chevron Chemical Co. (1981) Results of Analysis for Water, Soil, Vegetation, Straw and Grain Treated with Bolero 10G in a Rice Environment. (Compilation; unpublished study received Jan 19, 1983 under 239-EX-78; CDL:249307-A)
124278	Chevron Chemical Co. (1981) Results of Analysis for Water, Soil, Vegetation, Straw and Grain Treated with Bolero 8 EC in a Rice Environment. (Compilation; unpublished study received Jan 24, 1983 under 239-EX-77; CDL:249348-A)
139052	Schaefer, C.; Miura, T.; Stewart, R.; et al. (1981) Studies on the Potential Environmental Impact of the Herbicide Thiobencarb (Bolero). (Unpublished study received Feb 28, 1984 under 239-2450; prepared by Univ. of California--Fresno, Mosquito Control Re- search Laboratory, submitted by Chevron Chemical Co., Richmond, CA; CDL:252526-G)
140390	International Minerals & Chemical Corp. (1971) Description of the Analytical Method and Results of Tests on the Amount of Residue Remaining in Rice, Soil and Water: ?S-(4-Chlorobenzyl)-N,N-di-ethylthiolcarbamate|. (Compilation; unpublished study received Aug 3, 1972 under 2G1231; CDL:091083-H)
155451	Breault, G. (1985) Field Dissipation of Thiobencarb in Florida Muck Soil: Chevron File No. 741.11. Unpublished study prepared by Chevron Chemical Co. 17 p.
42003404	Lai, J. (1991) Field Dissipation of Bolero 8EC in Rice: Lab Project Number: 1641/90/7538. Unpublished study prepared by Chevron Chemical Co. 411 p.

164-2 Aquatic field dissipation

MRID	Citation Reference
25179	Lauck, J.E. (1979) Final Report of Field Study: Ortho Bolero 8 EC-- Rice, 1979. (Unpublished study received Dec 11, 1979 under 239- 2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-F)
25182	Serat, W.F.; Howard, J.; Cooper, W.D.; et al. (1975) Experimental Use Permit Health Effects: ?Bolero|. (Unpublished study received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-I)
33097	Chevron Chemical Company (1979) Environmental Chemistry: Summary: Bolero|. Summary of studies 241498-B and 241498-D through 241498-G. (Unpublished study received Dec 11, 1979 under 239- 2450; CDL:241498-A)
33098	Serat, W.F. (1975) Work Unit D--Experimental Use Permit Health Effects: Bolero|. (Unpublished study received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241498-B)
33753	Chevron Chemical Company (1979) Details of Application of Ortho Bolero 8 EC Herbicide under the EPA Experimental Use Permit No. 239-EUP-77 to Rice in the Chocolate Bayou Area of Texas. 1st inter. rept. (Unpublished study received Jan 15, 1980 under 239-EX-77; CDL:241723-A)
33756	Kincaide, R.T.; Slagowski, J.L.; Elliott, E.J.; et al. (1979) Interim Report Summary of Field Test at Arkansas Rice Experiment Station with Bolero 8 EC (Thiobencarb) Herbicide. (Unpublished study received Jan 15, 1980 under 239-EX-77; prepared in cooperation with Univ. of Arkansas, Rice Branch Experiment Station, submitted by Chevron Chemical Co., Richmond, Calif.; CDL: 241723-H)
72530	Serat, W.F. (1975) Work Unit D--Experimental Use Permit Health Effects: Bolero in Rice|. (Unpublished study received Dec 11, 1979 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241499-B)
72531	Kincade, R.T.; Smith, R. (1979) Interim Report Summary of Field Test at Arkansas Rice Experiment Station with Bolero 8 EC (Thiobencarb) Herbicide under EPA Permit No. 239-EUP-77. (Unpublished study received Dec 11, 1979 under 239-2449; prepared in cooperation with Univ. of Arkansas, Rice Branch Experiment Station, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241499-F)
82157	Kincaide, R.T.; Slagowski, J.L.; Elliott, E.J.; et al. (1979) Interim Report Summary of Field Test at Arkansas Rice Experiment Station with Bolero & EC (Thiobencarb) Herbicide under EPA Permit No. 239-EUP-77. (Unpublished study received Dec 11, 1979 under 239-2449; prepared in cooperation with Univ. of Arkansas, Rice Branch Experiment Station, submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241484-E)
124278	Chevron Chemical Co. (1981) Results of Analysis for Water, Soil, Vegetation, Straw and Grain Treated with Bolero 8 EC in a Rice Environment. (Compilation; unpublished study received Jan 24, 1983 under 239-EX-77; CDL:249348-A)
135377	Chevron Chemical Co. (1975) Benthiocarb: Residues in Rice and Other Crops|. (Compilation; unpublished study received Mar 18, 1976 under 6F1763; CDL:098089-M)
139052	Schaefer, C.; Miura, T.; Stewart, R.; et al. (1981) Studies on the Potential Environmental Impact of the Herbicide Thiobencarb (Bolero). (Unpublished study received Feb 28, 1984 under 239-2450; prepared by Univ. of California--Fresno, Mosquito Control Re- search Laboratory, submitted by Chevron Chemical Co., Richmond, CA; CDL:252526-G)
40651304	Ross, L.; Sava, R.; Oshima, R. (1988) Environmental Fate of Selected Rice Herbicides

(Thiobencarb and Molinate) under Field Conditions: Laboratory Project ID: R and RA 88-4. Unpublished study prepared by State of California, Dept. of Food and Agriculture. 127 p.

40651317 Ross, L.; Sava, R. (1986) Fate of thiobencarb and molinate in rice fields. J. Environ. Qual. 15(3): 220-225.

40651322 Comacchia, J.; Cohen, D.; Bowles, G.; et al. (1988) Rice Herbicides: Molinate and Thiobencarb: A Water Quality Assessment: Laboratory Project ID: R and RA 88-26. Unpublished study prepared by State of California. 236 p.

42003404 Lai, J. (1991) Field Dissipation of Bolero 8EC in Rice: Lab Project Number: 1641/90/7538. Unpublished study prepared by Chevron Chemical Co. 411 p.

43404005 Ho, B. (1994) Aquatic Field Dissipation of BOLERO 10G in Rice: Lab Project Number: T-7230. Unpublished study prepared by Chevron Chemical Co. 303 p.

164-5 Long term soil dissipation

MRID	Citation Reference
152879	Chevron Chemical Co. (1985) Thiobencarb: Amendment to Residue Tolerance Petition 5F3158: Section D: Celery, Endive (Escarole), Lettuce. Unpublished compilation. 212 p.

165-4 Bioaccumulation in fish

MRID	Citation Reference
155428	Chevron Chemical Co. (1986) Bolero 8EC: Wildlife & Aquatic Organ- isms Data. Unpublished compilation. 680 p.
165066	Biospherics Inc. (1983) Bolero: Texas Biological Monitoring Study April through December 1983: Progress Report. Unpublished compilation. 26 p.
42460401	Thacker, J.; Strauss, K.; Smith, G. (1992) Thiobencarb: A Metabolic Fate Study with the Bluegill (Lepomis macrochirus): Lab Project Number: 263E-101. Unpublished study prepared by Wildlife International Ltd. 165 p.

165-5 Bioaccum-aquatic non-target

MRID	Citation Reference
41215306	Watanabe, S. (1985) Accumulation and excretion of herbicides in various tissues of mussel. ? 26(5):496-499.
42130705	Fujie, G. (1985) Addendum to Impact of Bolero Run-off on a Brackish Water Ecosystem in Matagorda, Texas III. Second Treatment Year (Third Study Year): Chevron Chemical Co. S-2132; Biospherics Project 382; RM/16A/3S/2. Unpublished study prepared by Chevron Chemical Co. 33 p.
42130706	Finlayson, B.; Lew, T. (1983) Rice Herbicide Concentration in Sacramento River and Associated Agricultural Drains, 1982: Environ- mental Services Branch Administrative Report 83-5. Unpublished study prepared by California Dept. of Fish and Game. 42 p.

MRID	Citation Reference
42130707	Finlayson, B.; Nelson, J.; Lew, T.; et al. (1982) Colusa Basin Drain and Reclamation Slough Monitoring Studies, 1980 and 1981: Environmental Services Branch Administrative Report 82-3. Unpublished study prepared by California Dept. of Fish and Game, Pesticides Investigations Unit. 61 p.
42130708	Fujie, G. (1983) Addendum to a Baseline Assessment to a Brackish Water Ecosystem, April 1, 1982 through March 31, 1983: Matagorda Texas, Chevron Chemical Co. S-2132: 382, S-2132/A1. Unpublished study prepared by Chevron Chemical Co. 25 p.
92182065	Manza, S. (1990) Chevron Chemical Company Phase 3 Summary of MRID 00133563 and Related MRIDs 00145833, 00145834, 00145835. Impact of Bolero Runoff on a Brackish Water Ecosystem: Project No. 382-1984. Prepared by Biospherics Inc. 14 p.

166-2 Ground water-small retrospective

MRID	Citation Reference
44650601	Okumura, D. (1998) (Thiobencarb and Other Chemicals: California Surface Water Monitoring Data: 1998 Rice Fields). Unpublished study prepared by California EPA, Dept. of Pesticide Regulation. 12 p.

850.1075 Fish acute toxicity test, freshwater and marine

MRID	Citation Reference
46091401	Albuquerque, R. (2003) Thiobencarb Technical: Acute Toxicity to Fish (Carp (*Cyprinus carpio*). Project Number: 200300280, KCI/236/032204, 17797. Unpublished study prepared by Huntingdon Life Sciences, Ltd. 38 p.

850.3020 Honey bee acute contact toxicity

MRID	Citation Reference
46059804	Wainwright, M. (2002) Thiobencarb Technical Acute Toxicity to Honey Bees (Apis mellifera). Project Number: 200300250, KCI/232/023980. Unpublished study prepared by Huntingdon Life Sciences, Ltd. 50 p.

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25175 or 138075	Kamienski, F.X. (1978) Evaluation of Hazard to Shrimp from Use of Bolero (Thiobencarb) 8EC for Rice Weed Control. (Unpublished study received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241490-A)
25780	Lauck, J.E. (1979) Observations on Avian Wildlife in Bolero Treated Rice Fields, Brazoria County, Texas. (Unpublished study received Dec 11, 1979 under 239-2450; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:241489-K)
33755	Lauck, J.E.; LeCompte, R.; Johnson, L.; et al. (1979) Observations on Avian Wildlife in Bolero Treated Rice Fields, Brazoria County, Texas. (Unpublished study received Jan 15, 1980 under 239-EX-77; prepared in cooperation with Chocolate Bayou Co., submit- ted by Chevron Chemical Co., Richmond, Calif.; CDL:241723-D)
39117	Warnock, R.E.; Cheng, H.M. (1974) Benthiocarb Metabolism in the Duck. (Unpublished study

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44510 Pack, D.E. (1976) Residues of Total Radioactivity in Daphnia magna Living in Water Treated with 14C-Benthiocarb: File No. 741.10. (Unpublished study received Mar 18, 1976 under 239-2449; submitted by Chevron Chemical Co., Richmond, Calif.; CDL:095095-R)

46059803 Rodgers, M. (2003) Thiobencarb Technical Acute Toxicity (LC 50) to the Earthworm. Project Number: 200300248, KCI/222/024088. Unpublished study prepared by Huntingdon Life Sciences, Ltd. 22 p.

46059805 Carter, J. (2003) Thiobencarb Effects on Soil Non-Target Micro-Organisms: Nitrogen Transformation: Carbon Transformation. Project Number: 200300251, KCI/237/032255. Unpublished study prepared by Huntingdon Life Sciences, Ltd. 57 p.

46091402 Albuquerque, R. (2003) Thiobencarb Technical Toxicity to the Sediment Dwelling Phase of the Midge Chironomus riparius. Project Number: 200300281, KCI/244/033011. Unpublished study prepared by Huntingdon Life Sciences, Ltd. 57 p.