

# **Water Quality Criteria Report for Bifenthrin**

## Phase III: Application of the pesticide water quality criteria methodology



Prepared for the Central Valley Regional Water Quality Control Board

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## **Disclaimer**

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## List of acronyms and abbreviations

ACR	Acute-to-Chronic Ratio
APHA	American Public Health Association
ASTM	American Society for Testing and Materials
BAF	Bioaccumulation Factor
BCF	Bioconcentration Factor
BMF	Biomagnification Factor
CAS	Chemical Abstract Service
CDFG	California Department of Fish and Game
CDPR	California Department of Pesticide Regulation
CDWR	California Department of Water Resources
CSIRO	Commonwealth Scientific and Industrial Research Organization, Australia
CVRWQCB	Central Valley Regional Water Quality Control Board
DOC	Dissolved Organic Carbon
DOM	Dissolved Organic Matter
EC <sub>x</sub>	Concentration that affects x% of exposed organisms
FDA	Food and Drug Administration
FT	Flow-through test
GMAV	Genus Mean Acute Value
IC <sub>x</sub>	Inhibition concentration; concentration causing x% inhibition
ICE	Interspecies Correlation Estimation
IUPAC	International Union of Pure and Applied Chemistry
K	Interaction Coefficient
K <sub>H</sub>	Henry's law constant
K <sub>ow</sub>	Octanol-Water partition coefficient
K <sub>p</sub> or K <sub>d</sub>	Solid-Water partition coefficient
LC <sub>x</sub>	Concentration lethal to x% of exposed organisms
LD <sub>x</sub>	Dose lethal to x% of exposed organisms
LL	Less relevant, Less reliable study
LOEC	Lowest-Observed Effect Concentration
LOEL	Lowest-Observed Effect Level
LR	Less relevant, Reliable study
MATC	Maximum Acceptable Toxicant Concentration
N	Not relevant or Not reliable study
n/a	Not applicable
NOAEL	No-Observed Adverse Effect Level
NOEC	No-Observed Effect Concentration
NR	Not reported
OC	Organic Carbon
OECD	Organization for Economic Co-operation and Development
PBO	Piperonyl butoxide
QSAR	Quantitative Structure Activity Relationship
pK <sub>a</sub>	Acid dissociation constant
RL	Relevant, Less reliable study
RR	Relevant and Reliable study

S	Static test
SMAV	Species Mean Acute Value
SMCV	Species Mean Chronic Value
SPME	Solid-phase Microextraction
SR	Static renewal test
SSD	Species Sensitivity Distribution
TES	Threatened and Endangered Species
TIE	Toxicity Identification Evaluation
US	United States
USEPA	United States Environmental Protection Agency

## 1. Introduction

A new methodology for deriving freshwater water quality criteria for the protection of aquatic life was developed by the University of California, Davis (TenBrook *et al.* 2009a). The need for a new methodology was identified by the California Central Valley Regional Water Quality Control Board (CVRWQCB 2006) and findings from a review of existing methodologies (TenBrook & Tjeerdema 2006, TenBrook *et al.* 2009b). This new methodology is currently being used to derive aquatic life criteria for several pesticides of particular concern in the Sacramento River and San Joaquin River watersheds. The methodology report (TenBrook *et al.* 2009a) contains an introduction (Chapter 1); the rationale of the selection of specific methods (Chapter 2); detailed procedures for criteria derivation (Chapter 3); and a chlorpyrifos criteria report (Chapter 4). This criteria report for bifenthrin describes, section by section, the procedures used to derive criteria according to the UC-Davis methodology. Also included are references to specific sections of the methodology procedures detailed in Chapter 3 of the report so that the reader can refer to the report for further details (TenBrook *et al.* 2009a).

## 2. Basic Information

Chemical: Bifenthrin (Fig. 1)

CAS: (2-methyl[1,1'-biphenyl]-3-yl)methyl (1*R*,3*R*)-rel-3-[(1*Z*)-2-chloro-3,3,3-trifluoro-1-propenyl]-2,2-dimethylcyclopropanecarboxylate

IUPAC: 2-methyl-3-phenylbenzyl (1*RS*)-cis-3-(2-chloro-3,3,3-trifluoroprop-1-enyl)-2,2-dimethylcyclopropanecarboxylate

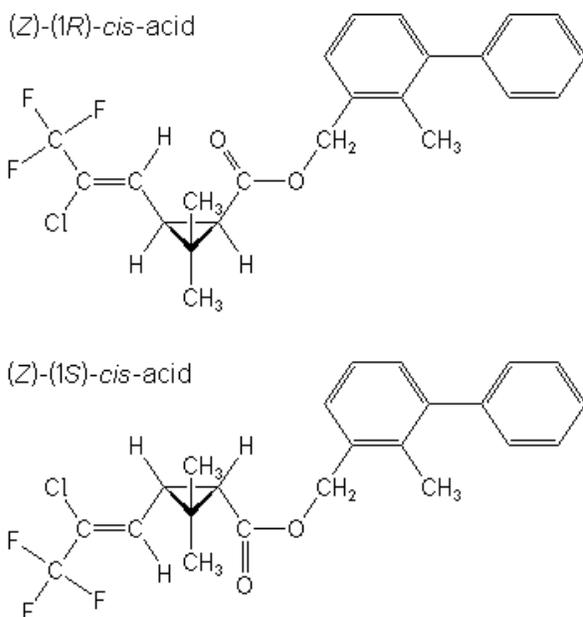


Figure 1. Structure of bifenthrin and stereoisomers (Wood 2008)

Chemical Formula: C<sub>23</sub>H<sub>22</sub>ClF<sub>3</sub>O<sub>2</sub>  
CAS Number: 82657-04-3  
CDPR Chem Code: 2300  
Classification: EPA Class C Carcinogen (EXTOXNET 1995)

Trade names: Bifenthrin, bifenthrine, Bifentrin, Bifentrina, Biflex, Biphenthrin, Brigade, Capture, Cyclopropanecarboxylic acid, FMC 54800, FMC 54800 Technical, Talstar, Tarstar, DeterMite, Biphenate, Torant (with Clofentezine), Zipak (with Amitraz) (EXTOXNET 1995, FMC Corp. 2007, Kegley *et al.* 2008)

### 3. Physical-Chemical Data

#### Molecular Weight

422.87 (EXTOXNET 1995, Laskowski 2002)

#### Density

1.26 g/mL (FOOTPRINT 2010)  
1.212 g/mL at 25°C (Meister 2002)  
Geomean: **1.24 g/mL**

#### Water Solubility

1 µg/L (Tomlin 2000)  
1 µg/L (FOOTPRINT 2010)  
Geomean: **1 µg/L**

#### Melting Point

Liquid at room temperature  
68-70.6 °C (EXTOXNET 1995)  
69.3 °C (FOOTPRINT 2010)  
Geomean: **69.3 °C**

#### Organic Carbon-Water Adsorption Coefficient (K<sub>oc</sub>)

6,314 (Kegley *et al.* 2008)  
237,000 (Laskowski 2002)  
380,000- 980,000 (Xu *et al.* 2007)  
236,610 (FOOTPRINT 2010)  
1.1x10<sup>5</sup> (9d equilibrium), 7.0 x 10<sup>5</sup> (30d equil.), both freshwater (Bondarenko *et al.* 2006)  
2.6 x 10<sup>5</sup> (9d equilibrium), 2.7 x 10<sup>5</sup> (30d equil.), both marine (Bondarenko *et al.* 2006)

#### Logistic Octanol-Water Partition Coefficient (Log K<sub>ow</sub>)

6.00 (Hansch *et al.* 1995, recommended by Sangster Research Laboratories 2007)  
5.56 using HPLC (Donovan & Pescatore 2002)  
6.4 (Laskowski 2002)  
7.3 at 20 °C calculated (FOOTPRINT 2010)  
Recommended: **6.00**

### Dissociation Coefficient ( $K_d$ )

390 (Surprenant 1988)

9,300- 18,900 (Xu *et al.* 2007)

1,400-15,100 (Yang *et al.* 2006a)

8,600-24,400 (Yang *et al.* 2006b)

### Vapor Pressure

1.80E-07 mm Hg at 25°C (Tomlin 1994, Laskowski 2002)

1.81E-07 mm Hg at 25°C (Meister 2002)

Geomean: **1.81E-07 mm Hg**

### Henry's Constant ( $K_H$ )

$7.2 \times 10^{-3} \text{ atm m}^3 \text{ mol}^{-1}$  (Laskowski 2002)

$7.74 \times 10^{-5} \text{ Pa m}^3 \text{ mol}^{-1}$ , at 25 °C (FOOTPRINT 2010)

$4.10 \times 10^{-2}$  dimensionless, at 20 °C (FOOTPRINT 2010)

### Bioconcentration Factors

Table 1. Bioconcentration factors (BCF) for bifenthrin; FT: flow-through; S: static; R: Recirculating. Values are on a wet weight basis and are not lipid normalized.

<b>Species</b>	<b>BCF</b>	<b>Exposure Type</b>	<b>Reference</b>
<i>Lepomis machrochirus</i> <sup>1</sup>	6090	FT, 42 d	Surprenant 1986
<i>Lepomis machrochirus</i> <sup>2</sup>	8720	FT, 42 d	Surprenant 1986
<i>Lepomis machrochirus</i> <sup>3</sup>	2140	FT, 42 d	Surprenant 1986
<i>Pimephales promelas</i> <sup>1</sup>	21,000-28,000	FT, Life Cycle	McAllister 1988
<i>Pimephales promelas</i> <sup>4</sup>	83-4900	FT, Life Cycle	McAllister 1988
<i>Pimephales promelas</i> <sup>5</sup>	530-10,000	FT, Life Cycle	McAllister 1988
<i>Pimephales promelas</i> <sup>6</sup>	6000	FT	McAllister 1988
<i>Pimephales promelas</i>	45-63	R, 21 d	Surprenant 1988
<i>Daphnia magna</i>	~ 1000-4600	S, 24 h	Yang <i>et al.</i> 2006a
<i>Daphnia magna</i> <sup>7</sup>	~ 1200-2600	S, 24 h, w/ sediment	Yang <i>et al.</i> 2006a
<i>Daphnia magna</i>	270-440	R, 21 d	Surprenant 1988
<i>Asellus sp.</i>	71-82	R, 21 d	Surprenant 1988
<i>Asellus sp.</i>	120-180	R, 21 d, w/ soil	Surprenant 1988
<i>Corbicula</i>	41-74	R, 21 d	Surprenant 1988
<i>Corbicula</i>	92-140	R, 21 d, w/ soil	Surprenant 1988

<sup>1</sup>whole body, <sup>2</sup>viscera, <sup>3</sup>fillet, <sup>4</sup><48h embryos, <sup>5</sup>96h embryos, <sup>6</sup>14d larvae, <sup>7</sup> with suspended solids (0-200 mg/L)

### Half-life

anaerobic soil degradation: 425 d (Laskowski 2002)

anaerobic soil degradation: 179.5 d (Kegley *et al.* 2008)

aerobic soil degradation: 96 d (Laskowski 2002)

aerobic soil degradation: 123.0 d (Kegley *et al.* 2008)

sediment: 8-17 mo at 20°C (Gan *et al.* 2005)

soils: 44-47 mo at 25°C (Baskaran *et al.* 1999)

hydrolysis: stable (Laskowski 2002)

photolysis, water: 408 d (Laskowski 2002)

photolysis, soil: 96.9 d (Laskowski 2002)

#### 4. Mode of Action and Toxicity

Pyrethroids affect the nervous system and induce paralysis in insects. More specifically, these compounds prevent sodium and potassium channels in the neuronal membranes from closing, causing over-excitation of neurons. The site of toxic action is very similar to that for DDT (Miller & Salgado 1985). Aquatic organisms are inherently more sensitive to pyrethroid pesticides than their terrestrial counterparts (Siegfried 1993), due to the effect of pyrethroids on Na<sup>+</sup> ATPase, an enzyme crucial to osmoregulation (Clark & Matsumura 1982).

Pyrethroids are chiral compounds consisting of multiple stereoisomers. The commercial formulations of bifenthrin are made up of 1*R*-*cis*-BF and 1*S*-*cis*-BF isomers (Figure 1). The 1*R*-*cis* enantiomer was the only enantiomer in *cis*-BF showing acute toxicity against *Ceriodaphnia dubia* (Liu *et al.* 2005a, b). Additionally, it was found that the 1*S*-*cis* enantiomer was preferentially degraded over the 1*R*-*cis* enantiomer, so the more toxic isomer was also more persistent in this case (Liu *et al.* 2005a, b).

In addition to acute toxicity, pyrethroids can induce sublethal toxicity such as altered behavior, reduced growth, immune system effects, endocrine reproductive effects, histopathological effects, as well as biochemical responses. Such sublethal effects may cause changes in predation avoidance, competition, learning and other characteristics that can affect survival and reproductive success (Werner & Moran 2008). Direct links of these effects to survival are difficult to establish. However, these effects likely contribute to negative effects on survival, growth, or reproduction, which are measured in standard chronic toxicity tests. Solomon *et al.* (2001) compiled toxicity data available for several pyrethroids and found acute-to-chronic ratios (ACRs) of 2 - 425 for pyrethroids in a variety of species. The large ACRs were not just for fish. Using the data for *Daphnia magna*, calculated ACRs for cypermethrin, tralomethrin, and λ-cyhalothrin were around 100, while those for cyfluthrin, fenvalerate/esfenvalerate, permethrin, and fenpropathrin were around 5. Chronic toxicity data for sensitive species is needed to derive fully protective criteria for pyrethroids.

#### 5. Environmental and Metabolic Fate

Bifenthrin, a third-generation synthetic pyrethroid, has greater photostability and enhanced insecticidal activity in comparison to older formulations (Mokry & Hoagland 1990). Bifenthrin is non-polar and has a strong affinity for soil particles and organic matter as represented by its high organic carbon (OC)-water adsorption partition coefficient ( $K_{OC}$ ; see section 3). The strong sorption to soils and the low water solubility would seem to confine these compounds to areas of use. However, they are able to move with runoff into surface streams by moving with suspended sediments and dissolved organic matter (DOM; Gan *et al.* 2005, Weston *et al.* 2004). The toxicity of pyrethroids to wildlife may be mitigated by their high affinity for suspended particulates (Hill 1989, Muir *et al.* 1985), and likewise toxicity during laboratory

testing may be reduced due to surface adherence (Froelich *et al.* 1984).

A study of bifenthrin and three other pyrethroids by Bondarenko *et al.* (2006), which examined the time-dependence of pyrethroids distributed in the freely dissolved, DOM, and solid phases, found only a small percentage of these compounds in the freely-dissolved portion of several samples. In addition, there was a significant difference between the amounts of freely-dissolved bifenthrin in the sample after 9 days, when compared with the same fraction after 30 days, suggesting that bifenthrin takes a long time to reach equilibrium within an aquatic system (Bondarenko *et al.* 2006).

Bifenthrin is stable in water and has a relatively long half-life in soils and sediments (see values in section 3). Long persistence was observed for bifenthrin under both aerobic and anaerobic conditions, and the half-life ranged from 8 to 17 months at 20 °C (Gan *et al.* 2005). Although pyrethroids are prone to cleavage at their ester linkage (Bradbury & Coats 1989, Tyler *et al.* 2000), upon binding to particulate matter the microbial degradation slows significantly and the half-life increases (Lee *et al.* 2004).

## **6. Human and Wildlife Dietary Values**

There are no FDA action levels for bifenthrin (USFDA 2000). There are no food tolerances for fish, but there are food tolerances for meat of cattle, goat, hogs, horses, and sheep at 0.5 ppm (USEPA 2006a).

### Wildlife toxicity values (dietary) for animals with significant food sources in water

For mallard ducklings, Fletcher (1983a) reported an eight day dietary LC<sub>50</sub> value of 1280 mg/kg feed. No ducklings died from the lowest dose, the 312 mg/kg feed, but these ducklings weighed less than the control ducklings. An acute study that monitored ducks for 21 days after a single dose of pure bifenthrin (not in feed) found no effects (Fletcher 1983b). Using the highest dose the NOEC would be 2150 mg/kg body weight for adult mallards (Fletcher 1983b). Roberts *et al.* (1986) observed no indication of reproductive impairment in mallards after they were fed a diet spiked with bifenthrin at three doses (25, 50, 75 mg/kg feed). Roberts *et al.* (1986) reported a NOEC of 75 mg/kg feed, but this likely an underestimated NOEC value because it is the highest dose and no toxicity was observed.

## **7. Ecotoxicity Data**

Approximately 40 original studies on the effects of bifenthrin on aquatic life were identified and reviewed. In the review process, many parameters are rated for documentation and acceptability for each study, including, but not limited to: organism source and care, control description and response, chemical purity, concentrations tested, water quality conditions, and statistical methods (see Tables 3.6, 3.7, 3.8 in TenBrook *et al.* 2009a). Single-species effects studies that were rated relevant (R) or

less relevant (L) according to the method were summarized in the data summary sheets. Information in these summaries was used to evaluate each study for reliability using the rating systems described in the methodology (Tables 3.7 and 3.8, section 3-2.2, TenBrook *et al.* 2009a), to give a reliability rating of reliable (R), less reliable (L), or not reliable (N). Copies of completed summaries for all studies are included in Appendix B of this report. Bifenthrin studies deemed irrelevant from an initial screening were not summarized (e.g., studies involving rodents or *in vitro* exposures). All data rated as acceptable (RR) or supplemental (RL, LR, LL) for criteria derivation are summarized in Tables 2 - 6, found at the end of this report. Acceptable studies rated as RR are used for numeric criteria derivation, while supplemental studies rated as RL, LR or LL are used for evaluation of the criteria to check that they are protective of particularly sensitive species and threatened and endangered species. These considerations are reviewed in sections 14 and 16 of this report, respectively. Studies that were rated not relevant (N) or not reliable (RN or LN) were not used for criteria derivation.

Using the data evaluation criteria (section 3-2.2, TenBrook *et al.* 2009a), nine acute toxicity studies, yielding twenty-one toxicity values from eight taxa, were judged reliable and relevant (RR; Tables 2 and 3). Two chronic toxicity studies, yielding four toxicity values from two taxa, were judged reliable and relevant (RR; Tables 4 and 5). Eleven studies were rated RL, LL, or LR and were used as supplemental information for evaluation of the derived criteria in Sections 14 and 16 (Table 6).

Eleven mesocosm, microcosm and ecosystem (field and laboratory) studies were identified and reviewed. Four of these studies were rated R or L and were used as supporting data in section 15 (Table 7). Three relevant studies of bifenthrin effects on wildlife were identified and reviewed for consideration of bioaccumulation in section 17.

## **8. Data Reduction**

Multiple toxicity values for bifenthrin for the same species were reduced into one species mean acute toxicity value (SMAV) or one species mean chronic value (SMCV) according to procedures described in the methodology (section 3-2.4, TenBrook *et al.* 2009a). Acceptable acute and chronic data that were reduced, and the reasons for their exclusion, are shown in Tables 3 and 5, respectively. Reasons for reduction of data included: more sensitive endpoints were available for the same test and more appropriate or more sensitive test durations were available for the same test. The final acute and chronic data sets are shown in Tables 2 and 4, respectively. The final acute data set contains eight SMAVs, and the final chronic data set contains two SMCVs.

## **9. Acute Criterion Calculation**

At least five acceptable acute toxicity values were available and fulfilled the five taxa requirements of the species sensitivity distribution (SSD) procedure (section

3-3.1, TenBrook *et al.* 2009a). The five taxa requirements are a warm water fish, a fish in the family Salmonidae, a planktonic crustacean, a benthic crustacean, and an insect. The log-logistic SSD procedure (section 3-3.2.2, TenBrook *et al.* 2009a) was used for the acute criterion calculation because there were not more than eight acceptable acute toxicity values available in the bifenthrin data set (Table 2). The log-logistic SSD procedure was used to derive 5<sup>th</sup> percentile values (median and lower 95% confidence limit), as well as 1<sup>st</sup> percentile values (median and lower 95% confidence limit). The median 5<sup>th</sup> percentile value is recommended for use in criteria derivation by the methodology because it is the most robust of the distributional estimates (section 3-3.2, TenBrook *et al.* 2009a). Comparing the median estimate to the lower 95% confidence limit of the 5<sup>th</sup> percentile values, it can be seen that the first significant figures of the two values are different (0.00803 vs. 0.000391 µg/L). Because there is uncertainty in the first significant digit, the final criterion will be reported with one significant digit (section 3-3.2.6, TenBrook *et al.* 2009a).

The ETX 1.3 Software program (Aldenberg 1993) was used to fit the a log-logistic distribution to the data set, which is plotted with the acute values in Figure 2. This distribution provided a satisfactory fit (see Appendix A) according to the fit test described in section 3-3.2.4 of TenBrook *et al.* (2009a). No significant lack of fit was found ( $\chi^2_{2n} = 0.2417$ ) using the fit test based on cross validation and Fisher's combined test (section 3-3.2.4, TenBrook *et al.* 2009a), indicating that the data set is valid for criteria derivation.

**Log-logistic distribution**

HC5 Fitting Parameter Estimates:  $\alpha = -0.661$ ,  $\beta$  (median) = 0.4872,  $\beta$  (lower 95% CI) = 0.9328.

- 5<sup>th</sup> percentile, 50% confidence limit: 0.00803 µg/L
- 5<sup>th</sup> percentile, 95% confidence limit: 0.000391 µg/L
- 1<sup>st</sup> percentile, 50% confidence limit: 0.00126 µg/L
- 1<sup>st</sup> percentile, 95% confidence limit: 0.0000113 µg/L

Recommended acute value = 0.00803 µg/L (median 5<sup>th</sup> percentile value)

Acute criterion = Recommended acute value ÷ 2  
 = 0.00803 µg/L ÷ 2  
 = 0.00402 µg/L

**Acute criterion** = 0.004 µg/L  
 = 4 ng/L

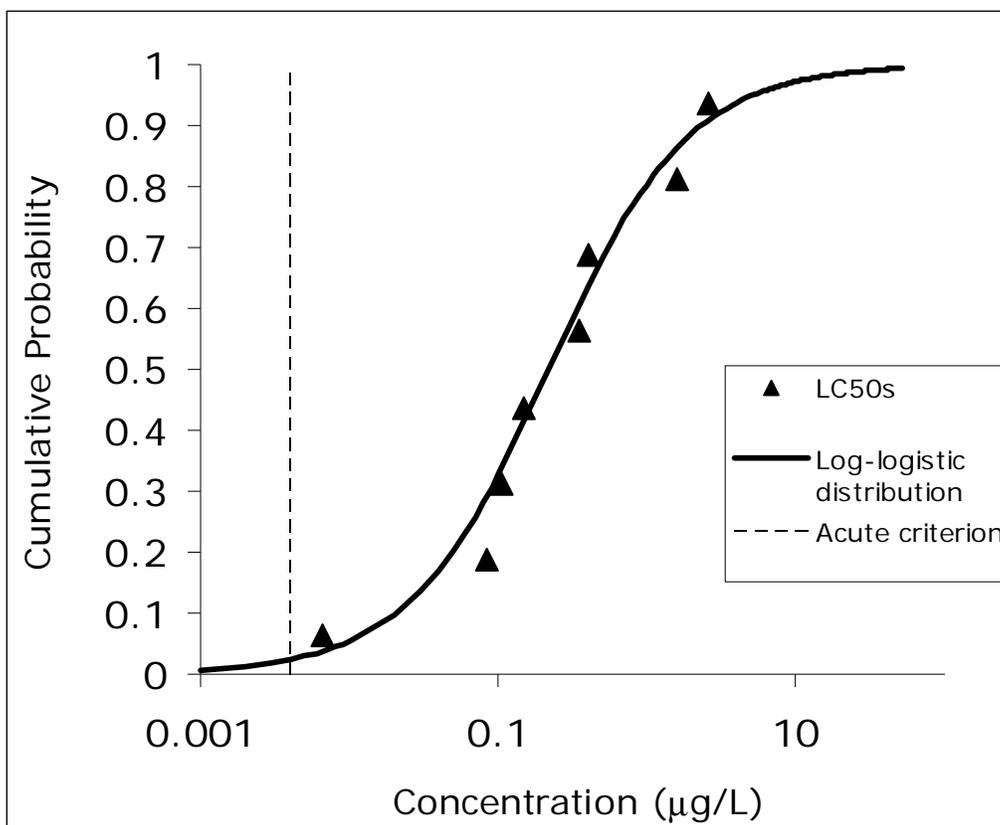


Figure 2. Bifenthrin acute data set fit to a log-logistic species sensitivity distribution.

## 10. Chronic Criterion Calculation

Chronic toxicity values from fewer than five different families were available, thus the ACR procedure was used to calculate the chronic criterion (section 3-4.2, TenBrook *et al.* 2009a). Two SMCVs are in the acceptable (rated RR) data set (Table 4), satisfying two of the five taxa requirements (section 3-3.1, TenBrook *et al.* 2009a): warm water fish (*Pimephales promelas*) and planktonic crustacean (*Daphnia magna*).

Neither of the above-mentioned chronic toxicity values could be paired with an appropriate corresponding acute toxicity value in order to calculate an ACR. The acute toxicity value for *Pimephales promelas* was conducted using a static test, which is inappropriate for determining a fish ACR (section 3-4.2.1, TenBrook *et al.* 2009a). For the *Daphnia magna* chronic toxicity value, there was another test that contained an acute toxicity value, but this test does not provide an appropriate corresponding value for an ACR because the test was not performed in the same laboratory or in the same dilution water (section 3-4.2.1, TenBrook *et al.* 2009a).

Salt-water data in the supplemental data set (Table 6) contained acute and chronic toxicity values for a mysid (*Americamysis bahia* – formerly *Mysidopsis bahia*), however the acute study was conducted in full seawater (30 ppt salinity), whereas the chronic studies were conducted in estuarine water (20 ppt salinity). These are not

appropriates corresponding toxicity values for an ACR, because the tests were not performed in the same dilution water (section 3-4.2.1, TenBrook *et al.* 2009a).

To avoid excessive layers of estimation, estimated chronic toxicity values using the Acute-to-chronic estimation software (ACE v. 2.0, USEPA 2003) were not derived to aid in calculating ACRs. Also, there were insufficient data to use this kind of estimation to produce chronic values for all five taxa that are required to construct a chronic SSD.

Because an ACR cannot be calculated with the available data, the chronic criterion was calculated with the default ACR value of 12.4 (section 3-4.2.3, TenBrook *et al.* 2009a). The chronic criterion was calculated using the recommended acute value and the default ACR value as follows:

$$\begin{aligned}\text{Chronic criterion} &= \text{Recommended acute value} \div \text{ACR} \\ &= 0.00803 \mu\text{g/L} \div 12.4 \\ &= 0.000648 \mu\text{g/L}\end{aligned}$$

$$\begin{aligned}\text{Chronic criterion} &= 0.0006 \mu\text{g/L} \\ &= 0.6 \text{ ng/L}\end{aligned}$$

## 11. Bioavailability

Although bifenthrin and other pyrethroids are not very soluble in water, aquatic organisms are very sensitive to pyrethroids and toxicity does occur. Several ecosystem and field studies are reviewed in section 15 that point to bifenthrin as the cause of toxicity in surface waters in the California Central Valley. This toxicity is believed to occur primarily from the fraction of the compound that is dissolved in the water, not from the compound that is associated with the particulate phase. Bioavailability of bifenthrin to organisms in the water column was demonstrated by Surprenant (1988). Bifenthrin from spiked soil samples was available at concentrations sufficient to cause toxicity to aquatic organisms (such as *Daphnia magna*) that were housed in a separate container from the sediment, but shared the same recirculating water (however, there was no filtration to prevent dissolved particles from moving, so particles could have been involved in the exposure).

Several studies suggest that the binding of bifenthrin to suspended solids and DOM will make the bound fraction unavailable and thus nontoxic to aquatic organisms. Yang *et al.* (2006a) found uptake of <sup>14</sup>C-labeled bifenthrin by *Daphnia magna* decreased with increasing suspended solids concentration, and that the organism uptake was closely mimicked by solid-phase microextraction (SPME) method using polydimethylsiloxane fibers. Regression analysis suggested that the portion of the pesticide sorbed to particles was unavailable to organisms in the 24-hr study period. In a complimentary study by Yang *et al.* (2006b), bifenthrin LC<sub>50</sub> values for *Ceriodaphnia dubia* were five times higher when 200 mg/L of suspended sediment was added compared to the sediment-free tests. Xu *et al.* (2007) tested bifenthrin toxicity to

*Chironomus tentans* in 10-d sediment exposures with three types of sediment. The researchers reported bifenthrin LC<sub>50</sub> values for five phases: bulk sediment, OC-normalized sediment, bulk porewater, dissolved organic carbon (DOC)-normalized porewater, and the freely dissolved bifenthrin. The LC<sub>50</sub> values in each of the five phases varied greatly, and varied between sediments for all phases tested except the freely dissolved, indicating that toxicity of the freely dissolved phase is independent of site-specific characteristics. The LC<sub>50</sub> values based on the freely dissolved concentrations (0.048-0.053 µg/L) were approximately an order of magnitude lower than those based on bulk porewater concentrations that included DOC (0.314-0.608 µg/L). These studies suggest that the freely dissolved concentration will be the most accurate predictor of toxicity and that bound bifenthrin was unavailable to the studied organisms.

As a counterpoint, equilibrium partitioning would suggest that as organisms take up bifenthrin, more bifenthrin will desorb from particles, so the fraction absorbed to solids is likely not completely unavailable. Although more bifenthrin could desorb from particles, the dissolved concentration should be constant if the system has reached a steady-state. Benthic organisms, such as *Hyaella azteca* may be at greater risk because of their exposure to porewater and close proximity to sediments.

Additionally, the role of dietary exposure on bioavailability of pyrethroids has not been considered. In the test with *Ceriodaphnia dubia* and *Daphnia magna*, organisms were not fed during the test duration (Yang *et al.* 2006a, 2006b). Organisms living in contaminated waters may also be ingesting food with sorbed hydrophobic compounds that can be desorbed by digestive juices (Mayer *et al.* 2001). The effects of dietary exposure may also be species-specific, depending on typical food sources; some species may have greater interaction with particles, increasing their exposure. Palmquist *et al.* (2008) examined the effects due to dietary exposure of the pyrethroid esfenvalerate on three aqueous insects with different feeding functions: a grazing scraper (*Cinygmula reticulata* McDunnough), an omnivore filter feeder (*Brachycentrus americanus* Banks), and a predator (*Hesperoperla pacifica* Banks). The researchers observed adverse effects in *C. reticulata* and *B. americanus* after feeding on esfenvalerate-laced food sources and that none of the three insects avoided the contaminated food. The effects included reduced growth and egg production of *C. reticulata* and abandonment and mortality in *B. americanus*. These limited studies indicate that ingestion may be an important exposure route, but it is not currently possible to incorporate this exposure route into criteria compliance assessment.

Section 3-5.1 of the methodology (TenBrook *et al.* 2009a) suggests that if studies indicate that fewer than three phases of the pesticide (sorbed to solids, sorbed to dissolved solids, or freely dissolved in the water) are bioavailable that compliance may be based on the concentration in the bioavailable phase(s). The studies above suggest that the freely dissolved fraction of bifenthrin is the primary bioavailable portion, and that this concentration is the best indicator of toxicity, thus, it is recommended that the freely dissolved fraction of bifenthrin be directly measured or calculated based on site-specific information for compliance assessment. Whole water concentrations are also

valid for criteria compliance assessment, and may be used at the discretion of environmental managers, although the bioavailable fraction may be overestimated with this method.

The most direct way to determine compliance would be to measure the bifenthrin concentration in the dissolved phase to determine the total bioavailable concentration. SPME has shown to be the best predictor of pyrethroid toxicity in several studies (Bondarenko *et al.* 2007, Bondarenko & Gan 2009, Hunter *et al.* 2008, Xu *et al.* 2007, Yang *et al.* 2006a, 2006b, 2007). Bondarenko & Gan (2009) report a method detection limit of 1.0 ng/L for bifenthrin, which is a factor of 4 below the acute criterion, and slightly higher than the chronic criterion. If method detection limits for the SPME method are not satisfactory compared to the criteria, this method may not be able to be used for criteria compliance; if detection limits of a given testing facility are shown to be satisfactory, the SPME method is valid for criteria compliance. Filtration of sediments is another option. Glass fiber filters with a nominal pore size of 0.7 µm or 0.45 µm are often used to remove the suspended sediments or both suspended sediments and DOM, but the filters can interfere with the detection of hydrophobic contaminants. Gomez-Gutierrez *et al.* (2007) found that adsorption to filters was positively correlated with the log  $K_{ow}$  and solubility values of the compounds, and that on average 58% of the one pyrethroid tested (a 50 ng/L solution of permethrin) was lost on the filter. This loss may be critical for determining compliance at environmental concentrations.

Alternately, the following equation can be used to translate total bifenthrin concentrations measured in whole water to the associated dissolved bifenthrin concentrations:

$$C_{dissolved} = \frac{C_{total}}{1 + ((K_{OC} \cdot [SS]) / f_{oc}) + (K_{DOC} \cdot [DOC])} \quad (1)$$

where:

- $C_{dissolved}$  = concentration of chemical in dissolved phase (µg/L);
- $C_{total}$  = total concentration of chemical in water (µg/L);
- $K_{OC}$  = OC-water partition coefficient (L/kg);
- [SS] = concentration of suspended solids in water (kg/L);
- $f_{oc}$  = fraction of OC in suspended sediment in water;
- [DOC] = concentration of dissolved organic carbon in water (kg/L);
- $K_{DOC}$  = OC-water partition coefficient (L/kg) for DOC.

To determine compliance by this calculation, site-specific data are necessary, including:  $K_{OC}$ ,  $K_{DOC}$ , the concentration of suspended solids, the concentration of DOC, and the fraction of OC in the suspended solids. If all of these site-specific data, including the partition coefficients, are not available, then this equation should not be used for compliance determination. Site-specific data are required because the sorption of bifenthrin to suspended solids and DOM depends on the physical and chemical properties of the suspended solids resulting in a range of  $K_{OC}$  values (see section 3).

The freely dissolved bifenthrin concentration is recommended for determination of criteria compliance because the literature suggests that the freely dissolved concentrations are the most accurate predictor of toxicity. Environmental managers may choose an appropriate method for determination of the concentration of freely dissolved bifenthrin, or they may also choose to base compliance on whole water concentrations.

## 12. Mixtures

Bifenthrin often occurs in the environment with other pyrethroid pesticides (Werner & Moran 2008). All pyrethroids have a similar mode of action, but some studies have indicated that pyrethroid mixture toxicities are not additive, and that slight antagonism can occur when pyrethroid mixture toxicity is tested. Definitions of additivity, synergism, antagonism, and non-additivity are available in the literature (Lydy and Austin 2004) and more detailed descriptions of mixture models can be found in the methodology (section 3-5.2, TenBrook *et al.* 2009a).

The effects on *Daphnia magna* mortality and feeding due to binary mixtures of lambda-cyhalothrin with deltamethrin, copper, and cadmium were examined in a study by Barata *et al.* (2006). The two concepts of concentration addition and independent action were used to predict mixture toxicity at various tested mixture ratios. Slight antagonism was observed in the lambda-cyhalothrin – deltamethrin mixture, which is unexpected because they have the same pharmacological mode of action. Neither method was able to consistently predict joint toxicity for the various mixtures. Brander *et al.* (2009) tested mixture toxicity of cyfluthrin and permethrin, and also found slight antagonism for the binary mixture, but additivity was demonstrated when piperonyl butoxide (PBO) was added. Brander *et al.* (2009) offered several explanations for the observed antagonism between the two pyrethroids. Permethrin is a type I pyrethroid, and cyfluthrin is a type II pyrethroid, and type II pyrethroids might be able to outcompete type I pyrethroids for binding sites, which is known as competitive agonism; or binding sites may be saturated, so that complete additivity is not observed. They also note that cyfluthrin is metabolized more slowly than permethrin, so cyfluthrin can bind longer. PBO may remove this effect because the rate of metabolism of both pyrethroids is reduced in the presence of PBO. The additivity of pyrethroid mixture toxicity has not been clearly defined in the literature, and in fact, antagonism has been observed, thus the concentration addition method is not recommended for use when multiple pyrethroids are found in a sample.

Piperonyl butoxide (PBO) is commonly added to pyrethroid insecticide treatments because it is known to increase the toxic effects of pyrethroids (Weston *et al.* 2006). Brander *et al.* (2009) observed *Hyalella azteca* LC<sub>50</sub> values decreased by a factor of 2 or 3.5 when a nonlethal concentration of PBO was mixed with cyfluthrin or permethrin, respectively. No interaction coefficients (K) have been derived with relevant species to describe synergism between bifenthrin and PBO. Consequently, it is not possible to quantify this non-additive toxicity and there is no accurate way to account for this interaction in compliance determination.

No studies on aquatic organisms were found in the literature that could provide a quantitative means to consider mixtures of bifenthrin with other classes of pesticides. However, several studies have been published that examine the interactive nature of bifenthrin with other pesticides and pesticide synergists in order to more effectively reduce a target pest or limit target insect resistance. The response of aquatic organisms, especially arthropods, may be comparable to the response of these targeted species (Werner & Moran 2008).

Several studies have used two similar methods to calculate the level of interaction between mixtures of bifenthrin. While their indexes do not provide a way to determine the toxicity of environmental mixtures, they provide information about the qualitative interaction. Bifenthrin toxicity to the diamondback moth (*Plutella xylostella*) was synergized by emamectin and spinosad, and were additive with those of chlorpyrifos and indoxacarb (Attique *et al.* 2006). Chlorpyrifos-methyl, another organophosphate pesticide, synergized effects of bifenthrin on the mosquito (*Anopheles gambiae*, Bonnet *et al.* 2004). Bifenthrin toxicity to the two-spotted spider mite (*Tetranychus urticae*) was synergized by acephate, amitraz, chlordimeform, profenofos, s,s,s-tributyl phosphorotrithionate, and dimethoate (Bynum *et al.* 1990, Bynum *et al.* 1997). In the Banks grass mite (*Oligonychus pratensis*) amitraz and s,s,s-tributyl phosphorotrithionate were synergistic (Bynum *et al.* 1997, Bynum & Archer 2002), while results with PBO varied from slightly synergistic to antagonistic (Bynum *et al.* 1997, Bynum & Archer 2002). It should also be noted that significant differences in response were observed between two closely related species tested in these studies (Bynum *et al.* 1997), which indicates that closely related aquatic organisms may also display a highly varied response to the same mixture of pesticides.

The silkworm, *Bombyx mori* (L.), a non-target organism, was exposed to leaves treated with a binary mixture of OP insecticides (dichlorvos and phoxim) and pyrethroid insecticides (permethrin, tetramethrin, bifenthrin, and ethofenprox), and experienced additive toxicity from the combination of pesticides (Zhang *et al.* 2008).

Although there are many examples of non-additive toxicity for bifenthrin and other chemicals, a multispecies interaction coefficient is not available for any chemical with bifenthrin, and therefore the concentrations of non-additive chemicals cannot be used for criteria compliance (section 3-5.2.2, TenBrook *et al.* 2009a).

### **13. Temperature, pH, and Other Water Quality Effects**

Temperature, pH, and other water quality effects on the toxicity of bifenthrin were examined to determine if any effects are described well enough in the literature to incorporate into criteria compliance (section 3-5.3, TenBrook *et al.* 2009a). Temperature has been found to be inversely proportional to the aquatic toxicity and bioavailability of pyrethroids (Miller & Salgado 1985, Werner & Moran 2008). In fact, the increase of toxicity of pyrethroids with decreasing temperature has been used to implicate pyrethroids as the source of toxicity in environmental samples (Phillips *et al.*

2004). The inverse relationship between temperature and pyrethroid toxicity is likely due to the increased sensitivity of an organism's sodium channels at low temperatures (Narahashi *et al.* 1998).

The toxicity of sediments contaminated with pyrethroids (often bifenthrin) was more than twice as toxic when tested at 18 °C compared to 23 °C (Weston *et al.* 2008). Weston *et al.* (2008) used a toxicity identification evaluation (TIE) procedure to determine the effect of temperature reduction (18 vs. 23 °C) on toxicity of a particular environmental sediment sample to *Hyalella azteca*. These results are not directly applicable for use in water quality criteria compliance because they were sediment exposures, and used environmental samples, instead of an exposure to a pure compound. This study does indicate that the enhanced toxic effects of pyrethroids at lower temperatures may not be as accurately represented by the results of typical laboratory toxicity tests, which tend to be run at warmer temperatures, 20-23 °C (USEPA 1996a, USEPA 1996b, USEPA 2000), than those of the habitats of coldwater fishes, about 15 °C or lower (Sullivan *et al.* 2000). In studies that used topical exposures (more relevant to spray application exposure to target a pest), the difference in toxicity can increase by a factor of about 1.5 to a factor of 10, in the temperature range of about 10 to 27 °C (Kumaraguru & Beamish 1981, Punzo 1993, Schnitzerling 1985).

Unfortunately, there are limited data using aquatic exposures with relevant species, making it unfeasible to quantify the relationship between the toxicity of bifenthrin and temperature for water quality criteria at this time (section 3-5.3, TenBrook *et al.* 2009a). No studies on bifenthrin were found that examined the effects of pH or other water quality parameters on toxicity, thus, there is no way to incorporate any of these parameters into criteria compliance.

#### 14. Sensitive Species

The derived criteria are compared to toxicity values for the most sensitive species in both the acceptable (RR) and supplemental (RL, LR, LL) data sets to ensure that these species will be adequately protected (section 3-6.1, TenBrook *et al.* 2009a). The lowest reported acute toxicity value in the RR data set (used directly in criteria calculation) is 2.7 ng/L for *Hyalella azteca* (Table 2), and the lowest species mean acute value in the RR data set is 6.5 ng/L for *H. azteca*. This value for *H. azteca* is the lowest compared to four others (9.3, 7.3, 8.0, 8.2 ng/L), although the 2.7 ng/L value is not considered an outlier. While there is one *H. azteca* toxicity value in the RR data set that is below the proposed acute criterion, the SMAV is the most robust toxicity value to represent a species. The *H. azteca* SMAV is based on five separate tests, and is therefore a more robust and reliable value than a single test value. A SMAV is calculated for use in the SSD so that no single species or single test for a species receives undue weight in the derivation process (section 2-2.7, TenBrook *et al.* 2009a). The goal of a SSD is to utilize the whole data set to derive protective estimates. In this case, it is not recommended that the acute criterion be adjusted downward based on one toxicity value for *H. azteca*, because the SMAV indicates that the acute criterion of 4

ng/L will be protective of this species. Downward adjustment of criteria can be recommended when a proposed criterion is higher than toxicity values for a sensitive species (section 3-6.1, TenBrook *et al.* 2009a), especially when there is very little data for a species, but it is not recommended in this case because there is ample highly rated data for *H. azteca*.

The lowest reported toxicity value in the supplemental data set (rated RL, LR, or LL, data not used directly in criteria calculation) is a LC<sub>50</sub> of 3.97 ng/L for *Americamysis bahia* (formerly *Mysidopsis bahia*) (Table 6), which is slightly below the acute criterion. The values for mysid are in the supplemental category because they are saltwater values, which may or may not be similar to toxicity values in freshwater. Saltwater data are not appropriate for use in criteria derivation or adjustment, but can be used for calculation of ACRs; thus downward adjustment of the acute criterion is not recommended based on this data.

The calculated chronic bifenthrin criterion (0.6 ng/L) is below the lowest chronic freshwater toxicity values in the data set. The lowest reported bifenthrin chronic toxicity value in the highly rated (RR) data set is a maximum acceptable toxicant concentration (MATC) of 1.9 ng/L for *Daphnia magna* (Table 3). In the supplemental data set, there is a chronic toxicity value of 1.25 ng/L for *Americamysis bahia* (formerly *Mysidopsis bahia*) (Table 6). The chronic criterion of 0.6 ng/L is approximately a factor of 3 and 2 below the *Daphnia magna* and *Mysidopsis bahia* values, respectively.

## 15. Ecosystem and Other Studies

The derived criteria are compared to acceptable laboratory, field, or semi-field multispecies studies (rated R or L) to determine if the criteria will be protective of ecosystems (section 3-6.2, TenBrook *et al.* 2009a). Eleven mesocosm, microcosm and ecosystem (field and laboratory) studies were identified and rated for reliability according to the methodology (Table 3.9, TenBrook *et al.* 2009a). Four of these studies were rated as reliable (R) or less reliable (L); all of the studies rated R or L are listed in Table 7. Some of the studies that rated as not reliable (N) are not discussed in this report (Giddings *et al.* 2001, Hendley *et al.* 2001, Maund *et al.* 2001, Travis & Hendley 2001). Several bifenthrin mesocosm tests were carried out with bifenthrin in the sediments, but bifenthrin was also measured in the water column. These studies simulate real world conditions, in which most of the bifenthrin would likely be bound to sediment.

Hoagland *et al.* (1993) examined the effects of sediment-associated bifenthrin alone and in combination with atrazine using tanks containing natural plankton assemblages and bluegill. The number of cladocerans (*Bosmina*), cyclopoid copepodids and copepods was reduced after 7 days at a concentration as low as 20 to 60 ng/L bifenthrin, while bluegill suffered 33% mortality at 3150 ng/L. Drenner *et al.* (1993) investigated the effect of sediment-associated bifenthrin on gizzard shad and plankton in outdoor tank mesocosms. Eight day LC<sub>50</sub> values for gizzard shad ranged from 207 -

521 ng/L (based on water concentrations 1 hour after sediment spiked with bifenthrin was added). In the same mesocosms, there was a significant decrease in copepod density and an increase in rotifer density.

Surprenant (1988) conducted experiments with soil that was spiked with 0.1 to 1 mg/kg bifenthrin in clean dilution water. Organisms were exposed to water only via circulation through different chambers for 21 days. *Daphnia magna* survival was significantly affected at 0.59 µg/L of bifenthrin. Survival of *Asellus sp.* was affected at bifenthrin concentrations of 0.30 µg/L and above. No toxic effects were seen in *Pimephales promelas* at 1.86 µg/L in water, and no toxic effects were seen in *Corbicula sp.* at 2.58 µg/L and below.

In these three studies (Drenner *et al.* 1993, Hoagland *et al.* 1993, Surprenant 1988) the toxic effects reported are all from concentrations above the derived bifenthrin chronic criterion of 0.6 ng/L. Based on these ecosystem studies, there is no evidence that the criteria will be underprotective of aquatic ecosystems.

To assess possible effects of bifenthrin field applications, Sherman (1989) documented extensive surveys of the aquatic organisms in two experimental ponds from 1986-1988, as well as *in situ* bioassays using *Daphnia magna* and *Pimephales promelas* exposures to spray drift and runoff. In the summer of 1986, ten weekly applications of a commercial formulation of bifenthrin, Capture 2.0, were sprayed on to agricultural fields at a rate of twice the then current label maximum (0.1 lbs/acre). These fields drained into nearby Hagan's Pond, which was a little over 3 acres in size. Observed toxic effects were compared to data from a reference pond 19 km to the north. The post application follow-up studies continued through August of 1987 and again in the summer of 1988, monitoring for recovery.

Of the zooplankton, calanoid copepods were clearly affected, while cladocerans showed some bifenthrin related effects. The survival and reproduction of ramshorn snail were negatively affected. Macroinvertebrates reduced in both density and number, but showed recovery. The bioassays with *Daphnia magna* and *Pimephales promelas* showed significant toxic effects and recovery. Phytoplankton, caged shrimp and crayfish exposed showed no clear effects. Mussels were unaffected and fish suffered no acute effects. There was a gizzard shad die off in the winter of 1987-88, but this seems to have not been bifenthrin related, as it did not correlate well to high concentrations of bifenthrin. Unfortunately the concentrations of bifenthrin cannot be directly tied to the observed effects. Average pond concentrations fluctuated from slightly above 1 ng/L to almost 10 ng/L from the summer of application until the next summer. The highest concentrations occurred in the summer of treatment, but overall there was not a clear temporal pattern as high concentrations were also observed in February and March of 1987, even though spraying ended in August of 1986 (see also Figure 1 in Palmieri 1988). The report also notes that herbicides and fertilizers were also applied during the study period. Since the concentrations that caused toxicity are not clear, this study cannot be used to judge if the derived criteria will be protective.

Several recent studies on the toxicity of pyrethroid mixtures, inclusive of bifenthrin, have been performed by Donald Weston and colleagues at the University of California, Berkeley. These studies do not rate as high quality field or mesocosm studies by the methodology (section 3-6.2 and Table 3.9, TenBrook *et al.* 2009a) because they are not controlled exposures, but use environmental samples that could contain many chemicals. However, these studies are summarized here because they provide evidence that bifenthrin is bioavailable and present at concentrations toxic to aquatic life in several areas of the California Central Valley. They also utilize TIEs that use several lines of evidence to identify the agents causing toxicity in samples, and the methodology does not have a rating scheme or parameter for TIE data.

Weston *et al.* (2005) collected sediments from creeks near residential areas of Roseville, CA. Almost half of the sampled sites (9 of 21), caused >90 % mortality to the *Hyaella azteca*. Bifenthrin, a common ingredient in lawn-care products, was implicated as the primary cause of toxicity, followed by cyfluthrin and cypermethrin. Another study, performed in 2006, confirmed that residential high pyrethroid use, particularly of bifenthrin, was causing significant toxicity in urban creeks. This study found that most samples collected from creeks in a variety of Sacramento area locations were lethal to *Hyaella azteca* in lab tests, while the highest mortality occurred in samples from housing subdivisions (Amweg *et al.* 2006). Bifenthrin has also been implicated in toxicity in creeks that catch agriculture runoff. Sediment samples collected from six sites along a six kilometer stretch of Del Puerto Creek all caused >70% mortality in toxicity tests with *Hyaella azteca*. Bifenthrin was identified as the primary contributor to toxicity in nearly all sites at which toxicity was observed (Weston *et al.* 2008). These results demonstrate toxicity at environmental concentrations, but unfortunately none of these studies included associated water concentrations of bifenthrin to compare with the derived criteria in this report.

## 16. Threatened and Endangered Species

The derived criteria are compared to measured toxicity values for threatened and endangered species (TES), as well as to predicted toxicity values for TES, to ensure that they will be protective of these species (section 3-6.3, TenBrook *et al.* 2009a). Current lists of state and federally listed threatened and endangered animal species in California were obtained from the California Department of Fish and Game web site (<http://www.dfg.ca.gov/biogeodata/cnddb/pdfs/TEAnimals.pdf>; CDFG 2008). Only one of the listed animals is represented in the acute or chronic toxicity data set, steelhead trout (*Oncorhynchus mykiss*), with an LC<sub>50</sub> of 0.15 µg/L. No threatened or endangered species are listed in the supplemental data set (Table 6).

Some of the listed species are represented in the acute toxicity data set by members of the same family or genus. *Oncorhynchus mykiss* and *Pimephales promelas* can serve as surrogates in estimates for other species in the same family using the USEPA interspecies correlation estimation website (WEB-ICE v. 2.0; Raimondo *et al.* 2007). Unfortunately, the bifenthrin toxicity values were out of range of the values used to develop the model for most of the available species. Only a value of 0.252 µg/L

could be estimated for Coho salmon (*Oncorhynchus kisutch*). Other estimations could be made more generally for the families of Salmonidae and Cyprinidae. These estimates are 0.237 µg/L for Salmonidae to 0.307 µg/L for Cyprinidae and are shown with the listed endangered species of that family in Table 8.

No single species plant studies were found in the literature for use in criteria derivation, so no estimation could be made for plants on the state or federal endangered, threatened or rare species lists. In a pond study, phytoplankton were unaffected by bifenthrin (Sherman 1989). However, bifenthrin seemed to be beneficial in some instances and harmful in others, as reported in a mesocosm study that monitored primary productivity, green algae, chlorophyll, and other endpoints for photosynthetic organisms (Hoagland *et al.* 1993). Based on the mode of action, plants should be relatively insensitive to bifenthrin and the calculated bifenthrin criteria should be protective of aquatic plants.

The lowest toxicity value, from either experimental or estimated datasets, for a threatened or endangered species is the experimental LC<sub>50</sub> value of 0.15 µg/L for *Oncorhynchus mykiss* that was used in bifenthrin criteria derivation calculation. Therefore, based on the available data and the estimated values for animals, there is no evidence that the calculated acute and chronic bifenthrin criteria will be underprotective of threatened or endangered species. However, it is important to note that this assessment lacks chronic data and any data for crustaceans and insects, which would be the most sensitive species in the acute criterion data set for bifenthrin. No data were found for effects of bifenthrin on federally endangered crustaceans or insects, or acceptable surrogates (i.e., in the same family).

## 17. Bioaccumulation

Bioaccumulation was assessed to ensure that the derived criteria will not lead to unacceptable levels of bifenthrin in food items (section 3-7.1, TenBrook *et al.* 2009a). Bifenthrin has a mean log K<sub>ow</sub> of 6.0 and a molecular weight of 422.87 (section 3), which indicates its bioaccumulative potential (section 3-7.1, TenBrook *et al.* 2009a). No biomagnification factor (BMF) values were found in the literature for bifenthrin. Bioaccumulation of bifenthrin has been measured in several studies (Table 1), which are briefly summarized here. The bioconcentration Factor (BCF) in fish varied from 45 to 28,000 depending on the age of the fish and if the analysis was based on residues in the whole body or just the portion that a human might consume (fillet). A 1986 study that examined the elimination of bifenthrin from the bluegill found that it is very slowly eliminated from tissues. After 42 days of depuration, fish tissue concentrations of bifenthrin were reduced by about half (Surprenant 1986). A recent study with *Daphnia magna* found that the Bioaccumulation Factor (BAF) varies greatly with differing concentrations of suspended sediments. BAFs in *Daphnia magna* ranged from 1000 to 4,600. As the concentration of suspended sediments was increased (0-200 mg/L), the associated BAF values decreased to 1,000 to 2,600 times (Yang *et al.* 2006a).

To check that these criteria are protective of terrestrial wildlife that may consume aquatic organisms, a BAF will be used to estimate the water concentration that would roughly equate to a reported toxicity value for consumption of fish by terrestrial wildlife. These calculations are further described in section 3-7.1 of the methodology (TenBrook *et al.* 2009a). The BAF of a given chemical is the product of the BCF and a BMF, such that  $BAF = BCF * BMF$ . For a conservative estimate, the BCF value of 28,000 L/kg for whole fish will be used (McAllister 1988, Table 1). A default BMF value of 10 is used, based on the log  $K_{ow}$  of bifenthrin (Table 3.17, TenBrook *et al.* 2009a). An oral predator NOEC value of 75 mg/kg feed is used (Roberts *et al.* 1986), although toxicity was not observed at any of the three doses tested (25, 50, 75 mg/kg), making this likely an underestimated NOEC value. This dose will be used because there were effects seen at the lowest dose (312 mg/kg feed) in a mallard duckling study by Fletcher (1983a).

$$NOEC_{water} = \frac{NOEC_{oral\_predator}}{BCF_{food\_item} * BMF_{food\_item}}$$

Mallard: 
$$NOEC_{water} = \frac{75 \text{ mg/kg}}{28,000 \text{ L/kg} * 10} = 0.000267 \text{ mg/L} = 0.267 \text{ } \mu\text{g/L} = 267 \text{ ng/L}$$

To check that these criteria are protective of humans that may consume aquatic organisms, a BAF will be used to estimate the water concentration that would roughly equate to a limit for human food consumption. An appropriate BAF was not available in the data set. The BCF value of 2140 L/kg for fish fillet (Surprenant 1986, Table 1) and a default BMF are used to approximate a BAF. There are no tolerance or FDA action levels for fish tissue (USFDA 2000), but there are food tolerances for meat of cattle, goat, hogs, horses, and sheep at 0.5 ppm (USEPA 2006a). This value can be used to roughly estimate if bioconcentration could cause bifenthrin concentrations in fish tissues to be of concern to human health.

Human: 
$$NOEC_{water} = \frac{0.5 \text{ mg/kg}}{2,140 \text{ L/kg} * 10} = 0.0000234 \text{ mg/L} = 0.0234 \text{ } \mu\text{g/L} = 23 \text{ ng/L}$$

In this example, the derived chronic criterion of 0.6 ng/L is below the estimated water concentrations of concern for wildlife and humans by a factor of 445 and 38, respectively. Therefore, adhering to the derived bifenthrin criteria should not conflict with other efforts to protect wildlife or human health from bifenthrin exposure.

## 18. Harmonization with Air and Sediment Criteria

This section addresses how the maximum allowable concentration of bifenthrin might impact life in other environmental compartments through partitioning (section 3-7.2, TenBrook *et al.* 2009a). However, there are no federal or state sediment or air quality standards for bifenthrin (CARB 2005, CDWR 1995, USEPA 2006b, USEPA 2006c) to enable this kind of extrapolation. For biota, the limited data on bioconcentration or biomagnification of bifenthrin was addressed in the bioaccumulation section (section 17).

## **19. Assumptions, Limitations and Uncertainties**

The assumptions, limitations and uncertainties involved in criteria derivation should be available to inform environmental managers of the accuracy and confidence in the derived criteria (section 3-8.0, TenBrook *et al.* 2009a). Chapter 2 of the methodology discusses these points for each section as different procedures were chosen, such as the list of assumptions associated with using a SSD (section 2-3.1.5.1), and reviews the assumptions in section 2-7.0 (TenBrook *et al.* 2009a). This section summarizes any data limitations that affected the procedure used to determine the final bifenthrin criteria. The different calculations of distributional estimates included in section 9 of this report may be used to consider the uncertainty in the resulting acute criterion.

For bifenthrin, the major limitation was lack of data in the chronic toxicity data set. Three of five taxa requirements were not met (a salmonid, benthic crustacean and insect), which precluded the use of a SSD; therefore, an ACR was used to derive the chronic criterion. Since no acceptable ACRs were available for bifenthrin in the literature, the default value of 12.4 was used (as specified in section 3-4.2.3, TenBrook *et al.* 2009a). Particularly of concern for the chronic toxicity data set was the lack of data on *Hyalella azteca*, which was the most sensitive species in the acute toxicity data set. Uncertainty cannot be quantified for the chronic criterion because it was derived using an ACR, not an SSD.

Another concern that could not be accounted for quantitatively with the acute and chronic criteria is the increase in toxicity from lower temperatures. Most of the toxicity data were from tests performed at standard temperature, usually around 20 °C. However, many streams in the California Central Valley often have lower water temperatures. If colder water bodies are impacted by concentrations of bifenthrin, it may be appropriate to apply an additional safety factor to the bifenthrin criteria for those areas, to ensure adequate protection. A rough factor of two could be estimated from a study by Weston *et al.* (2008), however, a study relating temperature to toxicity of bifenthrin in multiple species, including *Hyalella azteca*, would be ideal to derive such an adjustment factor. We do not recommend an additional safety factor to account for temperature effects at this time, but environmental managers may want to consider this application if the criteria do not appear to be protective of organisms in a colder water body. If aquatic exposure data for multiple species demonstrating temperature effects becomes available in the future, a regression equation describing the effect should be incorporated into criteria compliance.

Although greater than additive effects have been observed for mixtures of pyrethroids and PBO, there is insufficient data to account for this interaction for compliance determination. This is a significant limitation because formulations that contain both pyrethroids and PBO are now available on the market. When additional highly rated data is available, the criteria should be recalculated to incorporate new research.

## 20. Comparison to National Standard Methods

This section is provided as a comparison between the UC-Davis methodology for criteria calculation (TenBrook *et al.* 2009a) and the current USEPA (1985) national standard. The following example bifenthrin criteria were generated using the USEPA 1985 methodology with the data set generated in this bifenthrin criteria report.

The USEPA acute methods have three additional taxa requirement beyond the five required by the SSD procedure of the UC-Davis methodology (section 3-3.1, TenBrook *et al.* 2009a). They are:

1. A third family in the phylum Chordata (e.g., fish, amphibian);
2. A family in a phylum other than Arthropoda or Chordata (e.g., Rotifera, Annelida, Mollusca);
3. A family in any order of insect or any phylum not already represented.

Two out of the three of these additional requirements are met as follows:

1. The other fish /amphibian requirement is met with data from fathead minnow.
2. This requirements not met because all data are from organisms in the phylum Arthropoda or Chordata.
3. This requirement is met because *Chironomus dilutus* (family: Diptera) is from a different family than *Procladius sp.* (family Ephemeroptera).

Strictly speaking, the USEPA methodology cannot be used to calculate an acute criterion for bifenthrin. However, since the California Department of Fish and Game have used data sets that met only seven of eight requirements in the USEPA methodology, this will be done here.

Using the log-triangular calculation (following the USEPA 1985 guidelines) and the bifenthrin data set from Table 2 containing eight species values, the following criterion was calculated (Note: USEPA methodology uses *genus* mean acute values, while *species* mean acute values are used in this methodology and are reported in Table 2. Since there is only one species from each genus in Table 2, this final data set would be the same in both schemes.):

Example Acute value (5<sup>th</sup> percentile value) = 0.0009543 µg/L

$$\begin{aligned}
\text{Example Acute Criterion} &= \text{acute value} \div 2 \\
&= 0.0009543 \mu\text{g/L} \div 2 = 0.0004772 \mu\text{g/L} \\
&= 0.00048 \mu\text{g/L} \\
&= 0.48 \text{ ng/L}
\end{aligned}$$

According to the USEPA (1985) method, the criterion is rounded to two significant digits. The example acute criterion derived according to the US EPA methodology is approximately an order of magnitude below the acute criterion derived using the UC-Davis methodology. The two methodologies use different distributions (log-triangular vs. log-logistic), which have been demonstrated to give different criteria results in UC-Davis chlorpyrifos and diazinon criteria reports (Palumbo *et al.* 2010, TenBrook *et al.* 2009a).

For the chronic criterion, the bifenthrin data set only has data from two species, which are not enough for use in a SSD by either method. The USEPA 1985 methodology contains a similar ACR procedure as in the methodology used in this criteria report, to be used when three acceptable ACRs are available. For cases in which three acceptable ACRs are not available, the USEPA methodology does not have a default ACR or alternative procedure. Since no acceptable ACR could be calculated with the bifenthrin data set, no chronic criterion can be calculated using the USEPA 1985 methodology.

## 21. Final Bifenthrin Criteria Statement

The final criteria statement is:

Aquatic life in the Sacramento River and San Joaquin River basins should not be affected unacceptably if the four-day average concentration of bifenthrin does not exceed 0.0006  $\mu\text{g/L}$  (0.6  $\text{ng/L}$ ) more than once every three years, on the average, and if the one-hour average concentration of bifenthrin does not exceed 0.004  $\mu\text{g/L}$  (4  $\text{ng/L}$ ) more than once every three years on the average.

Although the criteria were derived to be protective of aquatic life in the Sacramento and San Joaquin Rivers, these criteria would be appropriate for any freshwater ecosystem in North America, unless species more sensitive than are represented by the species examined in the development of these criteria are likely to occur in those ecosystems.

The final acute criterion was derived using the log-logistic SSD procedure (section 9) and the acute data used in criteria calculation are shown in Table 2. The chronic criterion was derived by use of a default ACR (section 10); chronic data rated RR are shown in Table 4.

To date, there are no USEPA water quality criteria or aquatic life benchmarks for bifenthrin. The California Department of Fish and Game (CDFG) composed a risk assessment report for synthetic pyrethroids (Siepmann & Holm 2000). CDFG

concluded that there was insufficient data to calculate criteria for bifenthrin using the USEPA (1985) methods. This report is concluded by reporting the lowest acute and chronic toxicity values found. The lowest genus mean acute value (GMAV) for bifenthrin was 3.97 ng/L for *Americamysis bahia* (formerly *Mysidopsis bahia*) and the lowest MATC was 60 ng/L for *Pimephales promelas*. The chronic criterion in this report is below the lowest chronic toxicity value from the CDFG report. The lowest acute toxicity value from the CDFG report is below the criteria derived here, but it is for a saltwater species which may be more sensitive than freshwater species. Solomon *et al.* (2001) performed a probabilistic risk assessment with pyrethroids. Saltwater and freshwater toxicity data were combined so the lowest toxicity value in the data set was 3.8 ng/L (for mysid, a saltwater species). The 5<sup>th</sup> percentile value for bifenthrin, based on a log-normal distribution, was also 3.8 ng/L, although much of the author's discussion centered on the 10<sup>th</sup> percentile as the protective limit, which was 15 ng/L for bifenthrin. For compounds that had larger toxicity data sets, separate analyses were performed for freshwater and saltwater data. Differences were found especially for invertebrates, which suggested that the risk to freshwater and saltwater organisms should be assessed separately.

The derived criteria appear to be protective considering bioaccumulation, ecosystem level toxicity and threatened and endangered species as discussed above in the report, but the criteria calculations should be updated whenever new data is available.

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## **Data Tables**

**Table 2. Final acute toxicity data set for bifenthrin.** All studies were rated Relevant and Reliable (RR) and were conducted at standard temperature. Values in bold are species mean acute values. Est: toxicity values were calculated based on estimated concentrations (calculated from the recovery of some concentrations), Meas: toxicity values were calculated based on measured concentrations, Nom: toxicity values were calculated based on nominal concentrations. S: static, SR: static renewal, FT: flow-through.

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	LC <sub>50</sub> /EC <sub>50</sub> (µg/L)	Reference
<i>Ceriodaphnia dubia</i>	Cladoceran	Daphniidae	SR	Est	97.8%	96 h	24.0-24.7	Mortality	<24 h	0.078	Guy 2000a
<i>Ceriodaphnia dubia</i>	Cladoceran	Daphniidae	S	Nom	97.0%	48 h	25	Mortality	<24 h	0.142	Wheelock et al. 2004
<i>Ceriodaphnia dubia</i>										<b>0.105</b>	GEOMEAN
<i>Chironomus dilutus</i> (formerly <i>C. tentans</i> )	Midge	Chironomidae	S	Nom	100.0%	96 h	23 ± 1	Mortality	3 <sup>rd</sup> instar	<b>2.615</b>	Anderson et al. 2006
<i>Daphnia magna</i>	Cladoceran	Daphniidae	FT	Nom	88.4%	48 h	20-21	Mortality	<24 h	<b>1.6</b>	Surprenant 1983 MRID 132537
<i>Hyalella azteca</i>	Amphipod	Hyalellidae	S	Nom	100.0%	96 h	23 ± 1	Mortality	7-14 d	0.0093	Anderson et al. 2006
<i>Hyalella azteca</i>	Amphipod	Hyalellidae	SR	Est	98%	96 h	23 ± 1	Mortality	7-14 d	0.0027	Weston & Jackson 2009
<i>Hyalella azteca</i>	Amphipod	Hyalellidae	SR	Est	98%	96 h	23 ± 1	Mortality	7-14 d	0.0073	Weston & Jackson 2009
<i>Hyalella azteca</i>	Amphipod	Hyalellidae	SR	Est	98%	96 h	23 ± 1	Mortality	7-14 d	0.0080	Weston & Jackson 2009
<i>Hyalella azteca</i>	Amphipod	Hyalellidae	SR	Est	98%	96 h	23 ± 1	Mortality	7-14 d	0.0082	Weston & Jackson 2009
<i>Hyalella azteca</i>										<b>0.0065</b>	GEOMEAN
<i>Lepomis macrochirus</i>	Bluegill	Centrarchidae	FT	Nom	88.4%	96 h	21-22	Mortality	2.5 g, 8 mm	<b>0.35</b>	Hoberg 1983a MRID 00132536

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	LC <sub>50</sub> /EC <sub>50</sub> (µg/L)	Reference
<i>Oncorhynchus mykiss</i>	Rainbow trout	Salmonidae	FT	Nom	88.4%	96 h	11-12	Mortality	1.0 g, 46 mm	<b>0.15</b>	Hoberg 1983b MRID 00132539
<i>Pimephales promelas</i>	Fathead minnow	Cyprinidae	S	Meas	96.2%	96 h	25 ± 1	Mortality	40 d, 0.059g	0.21	McAllister 1988 MRID 40791301
<i>Pimephales promelas</i>	Fathead minnow	Cyprinidae	SR	Est	97.8%	96 h	24.0-24.5	Mortality	8 d, 0.0039-0.0052g	0.78	Guy 2000b
<i>Pimephales promelas</i>										<b>0.405</b>	GEOMEAN
<i>Proclonon sp</i>	Mayfly	Baetidae	S	Nom	100.0%	48 h	23 ± 1	Mortality	0.5-1.0 cm	<b>0.0843</b>	Anderson et al. 2006

**Table 3. Acceptable acute toxicity data for bifenthrin excluded in data reduction process.** All studies were rated relevant and reliable (RR). S: static, FT: flow-through.

Species	Common Identifier	Family	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/size	LC <sub>50</sub> / EC <sub>50</sub> (µg/L)	Reference	Reason for exclusion
<i>Lepomis macrochirus</i>	Bluegill	Centrarchidae	FT	Nom	88.4%	48 h	21-22	Mortality	2.5 g, 58 mm	0.65	Hoberg 1983a MRID 132536	1
<i>Lepomis macrochirus</i>	Bluegill	Centrarchidae	FT	Nom	88.4%	72 h	21-22	Mortality	2.5 g, 58 mm	0.44	Hoberg 1983a MRID 132536	1
<i>Lepomis macrochirus</i>	Bluegill	Centrarchidae	FT	Nom	88.4%	144 h	21-22	Mortality	2.5 g, 58 mm	0.3	Hoberg 1983a MRID 132536	1
<i>Oncorhynchus mykiss</i>	Rainbow trout	Salmonidae	FT	Nom	88.4%	24 h	11-12	Mortality	1.0 g, 46 mm	6.2	Hoberg 1983b MRID 132539	1
<i>Oncorhynchus mykiss</i>	Rainbow trout	Salmonidae	FT	Nom	88.4%	48 h	11-12	Mortality	1.0 g, 46 mm	0.34	Hoberg 1983b MRID 132539	1
<i>Oncorhynchus mykiss</i>	Rainbow trout	Salmonidae	FT	Nom	88.4%	72 h	11-12	Mortality	1.0 g, 46 mm	0.2	Hoberg 1983b MRID 132539	1
<i>Oncorhynchus mykiss</i>	Rainbow trout	Salmonidae	FT	Nom	88.4%	120 h	11-12	Mortality	1.0 g, 46 mm	0.1	Hoberg 1983b MRID 132539	1

Reasons for exclusion

1. A more sensitive or more appropriate test duration was available for the same test.

**Table 4. Final chronic toxicity data set for bifenthrin.** All studies were rated relevant and reliable (RR). FT: flow-through.

Species	Common Identifier	Test type	Meas/ Nom	Chemical	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L)	Reference
<i>Daphnia magna</i>	Cladoceran	FT	Meas	97.0%	21 d	19-22	Reproduction	< 24 h	0.0013	0.0029	0.0019	Burgess 1989 MRID 41156501
<i>Pimephales promelas</i>	Fathead minnow	FT	Meas	96.2%	92 d	25	Mortality	< 48 h	0.040	0.090	0.060	McAllister 1988 MRID 40791301

**Table 5. Acceptable chronic toxicity data for bifenthrin excluded in data reduction process.** All studies were rated relevant and reliable (RR). FT: flow-through.

Species	Common Identifier	Test type	Meas/ Nom	Chemical	Duration	Temp (°C)	Endpoint	Age/size	NOEC (µg/L)	LOEC (µg/L)	MATC (µg/L)	Reference	Reason for exclusion
<i>Daphnia magna</i>	Cladoceran	FT	Meas	97.0%	21 d	19-22	Time to 1 <sup>st</sup> brood	< 24 h	0.0029	0.0076	0.0047	Burgess 1989	1
<i>Daphnia magna</i>	Cladoceran	FT	Meas	97.0%	21 d	19-22	Length	< 24 h	0.0029	0.0076	0.0047	Burgess 1989	1

Reasons for exclusion

1. More sensitive endpoint available from same test

**Table 6. Supplemental studies excluded from bifenthrin criteria derivation (rated less relevant and/or less reliable: RL, LR, or LL). S: static, FT: flow-through.**

Species	Common Identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC <sub>50</sub> /EC <sub>50</sub> (µg/L)	MATC (µg/L)	Reference	Rating/ Reason
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	88%	96 h	21.5-21.6	Mortality	< 24 h	0.00397	-----	Barrows 1986b MRID 470271039	LR 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.5-25.7	Survival, F1	< 24 h	-----	0.00125	Boeri & Ward 1991 MRID 42338801	LR 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.5-25.7	Reproduction, young per female	< 24 h	-----	0.00343	Boeri & Ward 1991 MRID 42338801	LR 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.5-25.7	Growth, F1 length	< 24 h	-----	0.00125	Boeri & Ward 1991 MRID 42338801	LR 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.1-25.8	Survival F1,	< 24 h	-----	0.0025	Ward & Boeri 1991 MRID 41640501	LR 2, 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.1-25.8	Young per female,	< 24 h	-----	0.0025	Ward & Boeri 1991 MRID 41640501	LR 2, 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.1-25.8	F1 length,	< 24 h	-----	0.0025	Ward & Boeri 1991 MRID 41640501	LR 1, 3
<i>Americamysis bahia</i>	Mysid shrimp	FT	Meas	96.5%	28 d	23.1-25.8	Sublethal effects	< 24 h	-----	0.0025	Ward & Boeri 1991 MRID 41640501	LR 2, 3
<i>Ceriodaphnia dubia</i>	Cladoceran	S	Nom	96%	96 h	20	Mortality	< 20 h	0.144	-----	Liu et al. 2005a, 2005b	RL 2, 5
<i>Ceriodaphnia dubia</i>	Cladoceran	S	Nom	98%	96 h	21	Mortality	< 24 h	0.05	-----	Yang et al. 2006b	RL 5
<i>Cheumatopsyche</i> spp. & <i>Hydropsyche</i> spp.	Caddisfly	S	Nom	94%	24 h	20	Mortality	Larvae	7.2	-----	Siegfried 1993	RL 5
<i>Crassostrea virginica</i>	Eastern oyster	FT	Meas	88%	96 h	24	Reduced shell growth	31-50 mm height	> 2.15	-----	Ward 1986a MRID 470271040	LR 3, 4
<i>Crassostrea virginica</i>	Eastern oyster	FT	Meas	88%	96 h	26	Reduced shell growth	36-50 mm height	> 99.7	-----	Ward 1986b MRID 40266501	LR 3, 4

Species	Common Identifier	Test type	Meas/ Nom	Chemical grade	Duration	Temp (°C)	Endpoint	Age/ size	LC <sub>50</sub> /EC <sub>50</sub> (µg/L)	MATC (µg/L)	Reference	Rating/ Reason
<i>Cyprinodon variegatus</i>	Sheepshead minnow	FT	Meas	88%	96 h	19.9-22.3	Survival	9 wk	17.8	-----	Barrows 1986a MRID 470271038	LR 3
<i>Daphnia magna</i>	Cladoceran	FT	Meas	10.4%	48 h	19-21	Survival	≤ 24 h	0.11	-----	Hoberg et al. 1985 MRID 40275401	LR 1
<i>Daphnia magna</i>	Cladoceran	FT	Meas	10.4%	21 d	19-21	Survival	≤ 24 h	-----	0.01929	Hoberg et al. 1985 MRID 40275401	LR 1
<i>Daphnia magna</i>	Cladoceran	FT	Meas	10.4%	21 d	19-21	Reproduction	≤ 24 h	-----	0.0014	Hoberg et al. 1985 MRID 40275401	LR 1
<i>Daphnia magna</i>	Cladoceran	SR	Nom	99.5%	21 d	22	# of young/female	<24 h	-----	0.014	Wang et al. 2009	RL 2, 5
<i>Daphnia magna</i>	Cladoceran	SR	Nom	99.5%	21 d	22	Average brood size	<24 h	-----	0.014	Wang et al. 2009	RL 2, 5
<i>Daphnia magna</i>	Cladoceran	SR	Nom	99.5%	21 d	22	# of first brood/female	<24 h	-----	0.014	Wang et al. 2009	RL 2, 5
<i>Daphnia magna</i>	Cladoceran	SR	Nom	99.5%	21 d	22	Days to first brood	<24 h	-----	0.028	Wang et al. 2009	RL 2, 5
<i>Daphnia magna</i>	Cladoceran	SR	Nom	99.5%	21 d	22	Longevity	<24 h	0.031	0.014	Wang et al. 2009	RL 2, 5
<i>Enallagma</i> spp. & <i>Ishmura</i> spp.	Damselfly	S	Nom	94%	24 h	20	Mortality	Nymph	1.1	-----	Siegfried 1993	RL 5
<i>Heptageniidae</i> spp.	Mayfly	S	Nom	94%	24 h	20	Mortality	Nymph	2.3	-----	Siegfried 1993	RL 2, 5
<i>Hydrophilus</i> spp.	Diving beetle	S	Nom	94%	24 h	20	Mortality	Adult	5.4	-----	Siegfried 1993	RL 5
<i>Simulium vittatum</i>	Blackfly	S	Nom	94%	24 h	20	Mortality	Larvae	1.3	-----	Siegfried 1993	RL 5

#### Reasons for Rating

1. Low chemical grade
2. Control response not reported or not acceptable

3. Not freshwater
4. No toxicity value calculated
5. Low reliability score

**Table 7. Acceptable multispecies field, semi-field, laboratory, microcosm, mesocosm studies; R= reliable; L= less reliable.**

<b>Reference</b>	<b>Habitat</b>	<b>Rating</b>
Drenner <i>et al.</i> (1993)	Outdoor tank mesocosm	R
Hoagland <i>et al.</i> (1993)	Outdoor tank mesocosm	R
Sherman (1989)	Outdoor ponds	R
Surprenant (1988)	Indoor laboratory microcosm	R

**Table 8. Laboratory bifenthrin LC<sub>50</sub> values for threatened or endangered species and predicted values, using WEB-ICE (Raimondo *et al.* 2007).**

Species	Common Name	Family	LC <sub>50</sub> (µg/L)	Surrogate
Lab determined values for endangered species				
<i>Oncorhynchus mykiss</i>	Steelhead	Salmonidae	0.15	None - experimental value
Predicted based on species specific model				
<i>Oncorhynchus kisutch</i>	Coho salmon	Salmonidae	0.252	<i>Oncorhynchus mykiss</i>
Predicted with the family based model for Salmonidae				
<i>Oncorhynchus clarki</i>	Coho salmon	Salmonidae	0.237	<i>Oncorhynchus mykiss</i>
<i>Oncorhynchus mykiss</i>	Steelhead	Salmonidae	0.237	<i>Oncorhynchus mykiss</i>
<i>Oncorhynchus tshawytscha</i>	Chinook salmon	Salmonidae	0.237	<i>Oncorhynchus mykiss</i>
Predicted with the family based model for Cyprinidae				
<i>Gila elegans</i>	Bonytail chub	Cyprinidae	0.307	<i>Pimephales promelas</i>
<i>Ptychocheilus lucius</i>	Colorado squawfish	Cyprinidae	0.307	<i>Pimephales promelas</i>

## **Appendix A**

Fit test calculations

Raw data and calculations for fit test for bifenthrin acute data

Bifenthrin all LC 50s	Omit one								
	1	2	3	4	5	6	7	8	
<b>0.0065</b>	0.0065	0.0065	0.0065	0.0065	0.0065	0.0065	0.0065		
<b>0.0843</b>	0.0843	0.0843	0.0843	0.0843	0.0843	0.0843		0.0843	
<b>0.105</b>	0.105	0.105	0.105	0.105	0.105		0.105	0.105	
<b>0.15</b>	0.15	0.15	0.15	0.15		0.15	0.15	0.15	
<b>0.35</b>	0.35	0.35	0.35		0.35	0.35	0.35	0.35	
<b>0.405</b>	0.405	0.405		0.405	0.405	0.405	0.405	0.405	
<b>1.6</b>	1.6		1.6	1.6	1.6	1.6	1.6	1.6	
<b>2.615</b>		2.615	2.615	2.615	2.615	2.615	2.615	2.615	
<b>Omitted point, xi:</b>	2.615	1.6	0.405	0.35	0.15	0.105	0.0843	0.0065	
<b>5th percentile</b>	0.00803	0.00751	0.00654	0.0058	0.00585	0.00656	0.00712	0.0076	0.0349
Log logistic Distribution									
<b>F-i(xi)</b>		94.1	88.91	64.31	61.04	41.24	33.23	28.57	0.6271
		0.941	0.8891	0.6431	0.6104	0.4124	0.3323	0.2857	0.00627
<b>1-F(xi)</b>		0.059	0.1109	0.3569	0.3896	0.5876	0.6677	0.7143	0.99373
<b>Min of F-i(xi) or 1-F(xi)</b>		0.059	0.1109	0.3569	0.3896	0.4124	0.3323	0.2857	0.00627
<b>pi =2(min)</b>		0.118	0.2218	0.7138	0.7792	0.8248	0.6646	0.5714	0.01254

pi-values	ln(pi-value)	Fisher test statistic	
		Sum of ln(pi)	$X^2_{2n}$
0.1180	-2.1371	19.5384	0.2417
0.2218	-1.5060		
0.7138	-0.3372		
0.7792	-0.2495		
0.8248	-0.1926		
0.6646	-0.4086		
0.5714	-0.5597		
0.0125	-4.3787		

0.24 is  $> 0.05$  so the distribution fits the bifenthrin acute data set

if  $p < 0.05$  significant lack of fit

if  $p > 0.05$  fit (no significant lack of fit)