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Comparative aquatic toxicity of the pyrethroid insecticide lambda-cyhalothrin and its resolved isomer gamma-cyhalothrin

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Abstract In this review we compare the sensitivity of a range of aquatic invertebrate and fish species to gamma-cyhalothrin (GCH), the insecticidally active enantiomer of the synthetic pyrethroid lambda-cyhalothrin (LCH), in single-species laboratory tests and outdoor multi-species ecosystem tests. Species sensitivity distribution curves for GCH gave median HC_5 values of 0.47 ng/L for invertebrates, and 23.7 ng/L for fish, while curves for LCH gave median HC_5 values of 1.05 ng/L and 40.9 ng/L for invertebrates and fish, respectively. A model ecosystem test with GCH gave a community-level no observed effect concentration ($NOEC_{community}$) of 5 ng/L, while model ecosystem tests with LCH gave a $NOEC_{community}$ of 10 ng/L. These comparisons between GCH and LCH indicate that the single active enantiomer causes effects at approximately one-half the concentration at which the racemate causes similar effects.

Keywords Pyrethroid · Gamma-cyhalothrin · Lambda-cyhalothrin · Enantiomer · Species sensitivity distribution · Aquatic microcosm

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Introduction

Concerns have been raised about the reliability of environmental risk assessments for mixed enantiomer pyrethroid and organophosphate insecticides (Ali et al. 2003; Cai et al. 2008; Liu et al. 2004, 2005a, b). For example, Liu et al. (2005a) reported differences in toxicity to the aquatic invertebrates *Daphnia magna* and *Ceriodaphnia dubia* of 15- to 38-fold between enantiomers for bifenthrin and permethrin, and for cypermethrin and cyfluthrin only two of the eight isomers were toxic. In addition, for bifenthrin and permethrin Liu et al. (2005b) reported differences in degradation rates in sediments for different enantiomers and noted that such differences could significantly impact risk assessments. If toxicity is associated with a single enantiomer, ecotoxicological effects of enantiomer mixtures will depend upon the fate, behavior and bioavailability of only that enantiomer. To increase the accuracy and reliability of risk assessments for chiral pesticides, the toxicity and behavior of the component enantiomers need to be fully understood.

In this paper we will compare the extensive amount of aquatic toxicity data available for invertebrate and fish species for the synthetic pyrethroid lambda-cyhalothrin (LCH, CAS No. 91465-08-6) and its insecticidally active enantiomer gamma-cyhalothrin (GCH, CAS No. 76703-62-3). Lambda-cyhalothrin consists of two of the four enantiomers of the cyhalothrin molecule. Gamma-cyhalothrin is the insecticidally active enantiomer, 1R-cis, αS cyhalothrin. This enantiomer has been reported to be >162 times more toxic to zebrafish (*Danio rerio*) than 1R-cis, αR cyhalothrin, the other component of LCH (Xu et al. 2008). In this paper we compare data generated in laboratory studies with several fish and aquatic invertebrates to quantify the toxicological relationship between GCH and

LCH. The significance of these toxicological differences and of differences in environmental fate are further evaluated using data generated in multi-species outdoor micro/mesocosm studies under environmentally relevant conditions.

Single species laboratory studies

Data selection

Acute toxicity data for GCH (Table 1) were gleaned from studies meeting regulatory guidelines and using standard test species (*D. magna*, *Lepomis macrochirus*, *Oncorhynchus mykiss*, and *Selenastrum capricornutum*), guideline studies with additional fish species (*Poecilia reticulata*, *Brachydanio rerio*, and *Pimephales promelas*), a special study with three life stages of the amphipod *Gammarus pseudolimnaeus*, and another special study with nine additional aquatic invertebrate species (van Wijngaarden et al. 2008). Results of acute toxicity tests with LCH (Table 2) were found in the open literature (Maund et al. 1998; Schroer et al. 2004), the EPA Pesticide Toxicity Database (U.S. EPA 2007), and three unpublished guideline studies. Most of these studies used static or static-renewal exposures.

Many of the results for arthropods were reported as both immobilization (EC50s) and mortality (LC50s) after 48 and 96 h of exposure. In some of these tests immobilization was equal to mortality; that is, EC50s were the same as LC50s. In other tests, immobility did not lead to mortality, resulting in EC50s much lower than LC50s. Moreover, in many of the tests, some individuals that were immobile at 48 h had recovered when the 96-h observations were made. In such cases, 48-h EC50s were lower than 96-h EC50s, contrary to the general expectation that longer exposures lead to greater effects.

It can be argued that reversible sublethal effects are not of concern for acute risk assessments, and that lethal and sublethal endpoints should not be combined for analysis. Risk assessments for acute effects generally use toxicity data with mortality as the measured endpoint. For small invertebrates such as daphnids in which mortality can be difficult to confirm, immobilization is generally used as a surrogate for mortality. For example, the endpoints of the *G. pseudolimnaeus* tests were reported as "mortality/immobility" implying that immobilization was assumed equivalent to mortality. Use of sublethal EC50s for an acute risk assessment is a conservative (protective) approach.

Therefore, the invertebrate endpoints summarized in Tables 1 and 2 include EC50 and LC50 values, and in some cases for both 48 and 96 h. For fish, only the 96-h LC50 has been summarized and taken for the purpose of comparisons between GCH and LCH.

Comparison of toxicity data for individual species

The reported toxicity endpoints for GCH are presented in Table 1, and those for LCH in Table 2. Six arthropod species, five fish species, and one algal species were tested with both GCH and LCH (Table 3).

Of all the species tested only *Chaoborus obscuripes* was more sensitive to LCH than GCH. For the other arthropod species tested, the LCH/GCH ratio ranged from 1.05 (*Asellus aquaticus*) to 3.77 (*Cloeon dipterum*). The average ratio for the six arthropod species alone was 1.98. For fish, the LCH/GCH ratio ranged from 1.93 (*O. mykiss*) to 3.15 (*L. macrochirus*) for four species, consistent with GCH as a refined form of LCH, but was 13.5 for *P. reticulata*. The difference for *P. reticulata* was much greater than for any other species, a discrepancy that cannot be explained from the available information.

In addition to these exact comparisons, four pairs of closely related species were tested with GCH and LCH (Table 3). The LCH/GCH ratios for these four species were more variable than for the exact taxonomic matches, ranging from 0.40 to 1.53 for damselflies to 16.55 for chironomids.

Comparison of species sensitivity distributions

Invertebrates

Species sensitivity distributions (SSDs; see Posthuma et al. 2002) for GCH and LCH were generated based on 48-h EC50s and LC50s, 96-h EC50s and LC50s, and the lowest EC50s and LC50s for each invertebrate species (Fig. 1). The SSD describes the relationship between the concentration of GCH or LCH and the fraction of species whose EC50 or LC50 is exceeded at that concentration (the Potentially Affected Fraction). The SSD for the most sensitive endpoint, lowest EC50, is presented in Fig. 2 (upper left panel) with the relative positions of those species tested with both GCH and LCH identified.

To develop the SSDs, toxicity values (sample size $= n$) were first sorted from the lowest value to the highest value. Each value was assigned a rank and an associated probability of occurrence of the form $y_i = \text{rank}_i / n + 1$. A cumulative logistic regression function of the following form was fit to the data using the iteratively reweighted least squares maximum likelihood method shown in Eq. 1.

$$y_i = \frac{\exp(\alpha + \beta \ln(EC50_i))}{1 + \exp(\alpha + \beta \ln(EC50_i))} \quad (1)$$

Confidence intervals on y_i (95%) were calculated from the nonlinear model fit as a function of the parameter

Table 1 Toxicity of gamma-cyhalothrin to aquatic organisms

Scientific name	Common name	Time (h)	Endpoint	Measurement	Conc (ng/L)	Reference
Chaoborus obscuripes	Phantom midge	48	EC50	Immobilization	6.4	van Wijngaarden et al. (2008)
Chaoborus obscuripes	Phantom midge	48	LC50	Mortality	>36.1	van Wijngaarden et al. (2008)
Chaoborus obscuripes	Phantom midge	96	EC50	Immobilization	3.8	van Wijngaarden et al. (2008)
Chaoborus obscuripes	Phantom midge	96	LC50	Mortality	12.4	van Wijngaarden et al. (2008)
Chironomini	Midge	48	EC50	Immobilization	78.4	van Wijngaarden et al. (2008)
Chironomini	Midge	48	LC50	Mortality	>1,082	van Wijngaarden et al. (2008)
Chironomini	Midge	96	EC50	Immobilization	145	van Wijngaarden et al. (2008)
Chironomini	Midge	96	LC50	Mortality	>1,082	van Wijngaarden et al. (2008)
Cloeon dipterum	Mayfly	48	EC50	Immobilization	24.8	van Wijngaarden et al. (2008)
Cloeon dipterum	Mayfly	48	LC50	Mortality	887	van Wijngaarden et al. (2008)
Cloeon dipterum	Mayfly	96	EC50	Immobilization	23.4	van Wijngaarden et al. (2008)
Cloeon dipterum	Mayfly	96	LC50	Mortality	56.3	van Wijngaarden et al. (2008)
Corixa punctata	Water bug	48	EC50	Immobilization	12.3	van Wijngaarden et al. (2008)
Corixa punctata	Water bug	48	LC50	Mortality	64.6	van Wijngaarden et al. (2008)
Corixa punctata	Water bug	96	EC50	Immobilization	12.3	van Wijngaarden et al. (2008)
Corixa punctata	Water bug	96	LC50	Mortality	21.3	van Wijngaarden et al. (2008)
Notonecta maculata	Water bug	48	EC50	Immobilization	5.6	van Wijngaarden et al. (2008)
Notonecta maculata	Water bug	48	LC50	Mortality	65.7	van Wijngaarden et al. (2008)
Notonecta maculata	Water bug	96	EC50	Immobilization	4.6	van Wijngaarden et al. (2008)
Notonecta maculata	Water bug	96	LC50	Mortality	15.2	van Wijngaarden et al. (2008)
Zygoptera	Damselfly	48	EC50	Immobilization	304	van Wijngaarden et al. (2008
Zygoptera	Damselfly	48	LC50	Mortality	>3,610	van Wijngaarden et al. (2008)
Zygoptera	Damselfly	96	EC50	Immobilization	322	van Wijngaarden et al. (2008)
Zygoptera	Damselfly	96	LC50	Mortality	1,004	van Wijngaarden et al. (2008)
Asellus aquaticus	Isopod	48	EC50	Immobilization	26.2	van Wijngaarden et al. (2008)
Asellus aquaticus	Isopod	48	LC50	Mortality	253	van Wijngaarden et al. (2008)
Asellus aquaticus	Isopod	96	EC50	Immobilization	23.7	van Wijngaarden et al. (2008)
Asellus aquaticus	Isopod	96	LC50	Mortality	93.5	van Wijngaarden et al. (2008)
Daphnia magna	Water flea	48	EC50	Immobilization	45.0	Machado (2001b)
Daphnia magna	Water flea	48	EC50	Immobilization	99.4	Marino and Rick (2000b)
Daphnia magna	Water flea	504	NOEC	Reproduction	2.18	Kirk et al. (2001)
Gammarus pseudolimnaeus	Amphipod (adult)	48	EC50	Mortality/immobility	6.08	Henry et al. (2003a)
Gammarus pseudolimnaeus	Amphipod (adult)	48	EC50	Mortality/immobility	10.2	Henry et al. (2003b)
Gammarus pseudolimnaeus	Amphipod (juvenile)	48	EC50	Mortality/immobility	7.27	Henry et al. (2003b)
Gammarus pseudolimnaeus	Amphipod (neonate)	48	EC50	Mortality/immobility	0.894	Henry et al. (2003b)
Gammarus pseudolimnaeus	Amphipod (adult)	96	EC50	Mortality/immobility	3.05	Henry et al. (2003a)
Gammarus pseudolimnaeus	Amphipod (adult)	96	EC50	Mortality/immobility	6.32	Henry et al. (2003b)
Gammarus pseudolimnaeus	Amphipod (juvenile)	96	EC50	Mortality/immobility	3.58	Henry et al. (2003b)
Gammarus pseudolimnaeus	Amphipod (neonate)	96	EC50	Mortality/immobility	0.45	Henry et al. (2003b)
Gammarus pulex	Amphipod	48	EC50	Immobilization	5.0	van Wijngaarden et al. (2008)
Gammarus pulex	Amphipod	48	LC50	Mortality	16.1	van Wijngaarden et al. (2008)
Gammarus pulex	Amphipod	96	EC50	Immobilization	9.2	van Wijngaarden et al. (2008)
Gammarus pulex	Amphipod	96	LC50	Mortality	10.3	van Wijngaarden et al. (2008)
Proasellus coxalis	Isopod	48	EC50	Immobilization	17.7	van Wijngaarden et al. (2008)
Proasellus coxalis	Isopod	48	LC50	Mortality	218	van Wijngaarden et al. (2008)
Proasellus coxalis	Isopod	96	EC50	Immobilization	16.6	van Wijngaarden et al. (2008)
Proasellus coxalis	Isopod	96	LC50	Mortality	74.6	van Wijngaarden et al. (2008)
Brachydanio rerio	Zebrafish	96	LC50	Mortality	270	Sewell and McKenzie (2006c)

Table 1 continued

Scientific name	Common name	Time (h)	Endpoint	Measurement	Conc (ng/L)	Reference
Lepomis macrochirus	Bluegill	96	LC50	Mortality	63.1	Marino and Rick (2001b)
Lepomis macrochirus	Bluegill	96	LC50	Mortality	35.4	Marino and Rick (2001a)
Oncorhynchus mykiss	Rainbow trout	96	LC50	Mortality	170	Machado (2001a)
Oncorhynchus mykiss	Rainbow trout	96	LC50	Mortality	72	Marino and Rick (2000a)
Pimephales promelas	Fathead minnow	96	LC50	Mortality	340	Sewell and McKenzie (2006a)
Poecilia reticulata	Guppy	96	LC50	Mortality	170	Sewell and McKenzie (2006b)
Selenastrum capricornutum	Green alga	96	EC50	Growth	>1,340,000	Kirk et al. (2000)

Table 2 Toxicity of lambda-cyhalothrin to aquatic organisms

Scientific name	Common name	Time (h)	Endpoint	Measurement	Conc (ng/L)	Reference
Hydracarina	Water mite	48	EC50	Immobilization	47	Maund et al. (1998)
Asellus aquaticus	Isopod	48	EC50	Immobilization	26	Maund et al. (1998)
Asellus aquaticus	Isopod	48	EC50	Immobilization	24.8	Schroer et al. (2004)
Asellus aquaticus	Isopod	48	LC50	Mortality	140	Schroer et al. (2004)
Asellus aquaticus	Isopod	96	EC50	Immobilization	24.8	Schroer et al. (2004)
Asellus aquaticus	Isopod	96	LC50	Mortality	75.2	Schroer et al. (2004)
Cyclops sp.	Copepod	48	EC50	Immobilization	300	Maund et al. (1998)
Daphnia galeata	Water flea	48	EC50	Immobilization	117	Schroer et al. (2004)
Daphnia galeata	Water flea	48	LC50	Mortality	397	Schroer et al. (2004)
Daphnia magna	Water flea	48	EC50	Immobilization	362	Maund et al. (1998)
Daphnia magna	Water flea	48	EC50	Immobilization	51	Machado (2001b)
Daphnia magna	Water flea	504	NOEC	Reproduction	2.0	Maund et al. (1998)
Gammarus pulex	Amphipod	48	EC50	Immobilization	14	Maund et al. (1998)
Gammarus pulex	Amphipod	48	EC50	Not reported	6.8	U.S. EPA (2007)
Gammarus pulex	Amphipod	48	EC50	Immobilization	23.6	Schroer et al. (2004)
Gammarus pulex	Amphipod	48	LC50	Mortality	31.4	Schroer et al. (2004)
Gammarus pulex	Amphipod	96	EC50	Immobilization	24.2	Schroer et al. (2004)
Gammarus pulex	Amphipod	96	LC50	Mortality	24.2	Schroer et al. (2004)
Hyalella azteca	Amphipod	48	EC50	Immobilization	2.3	Maund et al. (1998)
Mysidopsis bahia	Mysid shrimp	96	LC50	Mortality	4.1	U.S. EPA (2007)
Ostracoda	Ostracod	48	EC50	Immobilization	3,300	Maund et al. (1998)
Proasellus coxalis	Isopod	48	EC50	Immobilization	17.7	Schroer et al. (2004)
Proasellus coxalis	Isopod	48	LC50	Mortality	78.8	Schroer et al. (2004)
Proasellus coxalis	Isopod	96	EC50	Immobilization	27.4	Schroer et al. (2004)
Proasellus coxalis	Isopod	96	LC50	Mortality	44.6	Schroer et al. (2004)
Simocephalus vetulus	Water flea	48	EC50	Immobilization	957	Schroer et al. (2004)
Simocephalus vetulus	Water flea	48	LC50	Mortality	1,340	Schroer et al. (2004)
Caenis horaria	Mayfly	48	EC50	Immobilization	17.9	Schroer et al. (2004)
Caenis horaria	Mayfly	48	LC50	Mortality	257	Schroer et al. (2004)
Caenis horaria	Mayfly	96	EC50	Immobilization	13.6	Schroer et al. (2004)
Caenis horaria	Mayfly	96	LC50	Mortality	34.6	Schroer et al. (2004)
Chaoborus obscuripes	Phantom midge	48	EC50	Immobilization	2.8	Schroer et al. (2004)
Chaoborus obscuripes	Phantom midge	48	LC50	Mortality	>27.4	Schroer et al. (2004)
Chaoborus obscuripes	Phantom midge	96	EC50	Immobilization	2.8	Schroer et al. (2004)
Chaoborus obscuripes	Phantom midge	96	LC50	Mortality	75.7	Schroer et al. (2004)
Chaoborus sp.	Phantom midge	48	EC50	Immobilization	2.8	Maund et al. (1998)
Chironomus riparius	Midge	48	EC50	Immobilization	2,400	Maund et al. (1998)

Table 2 continued

Scientific name	Common name	Time (h)	Endpoint	Measurement	Conc (ng/L)	Reference
Cloeon dipterum	Mayfly	48	EC50	Immobilization	38	Maund et al. (1998)
Cloeon dipterum	Mayfly	48	EC50	Immobilization	24.8	Schroer et al. (2004)
Cloeon dipterum	Mayfly	48	LC50	Mortality	122	Schroer et al. (2004)
Cloeon dipterum	Mayfly	96	EC50	Immobilization	88.3	Schroer et al. (2004)
Cloeon dipterum	Mayfly	96	LC50	Mortality	105	Schroer et al. (2004)
Corixa sp.	Water boatman	48	EC50	Immobilization	30	Maund et al. (1998)
Erythromma viridulum	Dragonfly	48	EC50	Immobilization	689	Schroer et al. (2004)
Erythromma viridulum	Dragonfly	48	LC50	Mortality	1,583	Schroer et al. (2004)
Erythromma viridulum	Dragonfly	96	EC50	Immobilization	493	Schroer et al. (2004)
Erythromma viridulum	Dragonfly	96	LC50	Mortality	493	Schroer et al. (2004)
Ischnura elegans	Damselfly	48	EC50	Immobilization	130	Maund et al. (1998)
Macropelopia sp.	Fly	48	EC50	Immobilization	244	Schroer et al. (2004)
Macropelopia sp.	Fly	48	LC50	Mortality	1,019	Schroer et al. (2004)
Macropelopia sp.	Fly	96	EC50	Immobilization	64.3	Schroer et al. (2004)
Macropelopia sp.	Fly	96	LC50	Mortality	698	Schroer et al. (2004)
Notonecta glauca	Water bug	48	EC50	Immobilization	14.8	Schroer et al. (2004)
Notonecta glauca	Water bug	48	LC50	Mortality	22.6	Schroer et al. (2004)
Notonecta glauca	Water bug	96	EC50	Immobilization	16.4	Schroer et al. (2004)
Notonecta glauca	Water bug	96	LC50	Mortality	16.4	Schroer et al. (2004)
Sialis lutaria	Alderfly	48	EC50	Immobilization	51.5	Schroer et al. (2004)
Sialis lutaria	Alderfly	48	LC50	Mortality	>2,179	Schroer et al. (2004)
Sialis lutaria	Alderfly	96	EC50	Immobilization	28	Schroer et al. (2004)
Sialis lutaria	Alderfly	96	LC50	Mortality	>2,179	Schroer et al. (2004)
Sigara striata	Water bug	48	EC50	Immobilization	18.2	Schroer et al. (2004)
Sigara striata	Water bug	48	LC50	Mortality	49.2	Schroer et al. (2004)
Brachydanio rerio	Zebra danio	96	LC50	Mortality	640	Maund et al. (1998)
Cyprinus carpio	Mirror carp	96	LC50	Mortality	500	Maund et al. (1998)
Gasterosteus aculeatus	Three-spined stickleback	96	LC50	Mortality	400	Maund et al. (1998)
Ictalurus punctatus	Channel catfish	96	LC50	Mortality	160	Maund et al. (1998)
Lepomis macrochirus	Bluegill	96	LC50	Mortality	210	Maund et al. (1998)
Lepomis macrochirus	Bluegill	96	LC50	Mortality	106	Marino and Rick (2001b)
Leuciscus idus	Golden orfe	96	LC50	Mortality	78	Maund et al. (1998)
Oncorhynchus mykiss	Rainbow trout	96	LC50	Mortality	240	Maund et al. (1998)
Oncorhynchus mykiss	Rainbow trout	96	LC50	Mortality	190	Machado (2001a)
Oryzias latipes	Japanese rice fish	96	LC50	Mortality	1,400	Maund et al. (1998)
Pimephales promelas	Fathead minnow	96	LC50	Mortality	700	Maund et al. (1998)
Poecilia reticulata	Guppy	96	LC50	Mortality	2,300	Maund et al. (1998)
Crassostrea gigas	Pacific oyster	48	EC50	Not reported	>590,000	U.S. EPA (2007)
Selenastrum capricornutum	Green alga	96	EC50	Growth	>1,000,000	Maund et al. (1998)

covariance matrix. The 50th percentile of the cumulative logistic function was estimated as exp $(-\alpha/\beta)$. The 5th percentile of the cumulative logistic function (the HC_5) was calculated as exp $[(-2.94 - \alpha)/\beta]$. The associated EC50 concentration units associated with the upper and lower confidence interval for any specific y_i were estimated through the equation by calculating the log $(EC50_i)$ associated with each bound.

The SSDs for GCH were approximately parallel to those for LCH, and always shifted slightly toward lower concentrations (Fig. 1). The differences in SSDs generated from different endpoints were small and mainly affected the SSD slopes. The HC_5 based on each species' lowest EC50 was 0.47 ng/L (95% confidence interval 0.11–2.00 ng/L) for GCH and 1.05 ng/L (0.47–2.34 ng/L) for LCH (Table 4).

Table 3 Comparison of acute toxicity of gamma-cyhalothrin (GCH) and lambda-cyhalothrin (LCH)

Species	GCH (ng/L)	LCH (ng/L)	LCH/GCH
Exact taxonomic match			
Chaoborus obscuripes (96-h EC50)	3.8	2.8	0.74
Gammarus pulex (96-h EC50)	9.2	24	2.63
Proasellus coxalis (96-h EC50)	17	27	1.65
Cloeon dipterum (96-h EC50)	23	88	3.77
Asellus aquaticus (96-h EC50)	24	25	1.05
Daphnia magna (48-h EC50, geometric mean, $N = 2$)	67	136	2.03
Lepomis macrochirus (96-h LC50, geometric mean, $N = 2$)	47	149	3.15
Oncorhynchus mykiss (96-h LC50, geometric mean, $N = 2$)	111	214	1.93
Poecilia reticulata (96-h LC50)	170	2,300	13.5
Brachydanio rerio (96-h LC50)	270	640	2.37
Pimephales promelas (96-h LC50)	340	700	2.06
Selenastrum capricornutum (96-h EC50)	>1,340,000	>1,000,000	NA[a]
Close taxonomic match			
Notonecta maculata (GCH), *N. glauca* (LCH) (96-h EC50)	4.6	15	3.33
Corixa punctata (GCH), *Corixa* sp. (LCH) (96-h EC50)	12	30	2.44
Chironomini (GCH), *C. riparius* (LCH) (96-h EC50)	145	2,400	16.55
Zygoptera (GCH), *Ischnura elegans* and *Erythromma viridulum* (LCH) (96-h EC50)	322	130 (*I. elegans*); 493 (*E. viridulum*)	0.40; 1.53
Arthropod species sensitivity distributions			
HC_5 (Lowest EC50)	0.47	1.05	2.23
HC_{50} (Lowest EC50)	14.6	48.5	3.32
Fish species sensitivity distributions			
HC_5 (Lowest LC50)	23.7	40.9	1.72
HC_{50} (Lowest LC50)	157.0	400.2	2.55

[a] *NA* Not applicable; cannot be calculated from available data

The SSD comparison based on the lowest EC50 values (Fig. 2, Table 4) supports two conclusions: (a) the rank order of species sensitivities is very similar for GCH and LCH, and (b) the entire SSD for GCH is shifted to the left of the SSD for LCH by a factor of approximately 2.2 at the HC_5 and 3.3 at the HC_{50}, similar to both the ratio of EC50s for individual arthropod species (Table 3) and the value expected based on isomeric composition.

The HC_5 of 0.47 ng/L for GCH differs from the value of 2.12 ng/L estimated by Van Wijngaarden et al. (2008). This difference can be ascribed to several factors. Most importantly, our SSD analysis included three species not included by those authors, notably *G. pseudolimnaeus* with an EC50 more than eight times lower than the next most sensitive species, *C. obscuripes*. Second, our SSD analysis was based on the lower of the 48- and 96-h EC50 values for each species, while van Wijngaarden et al. used only 96-h EC50 values. Third, the methods of HC_5 calculation differed between the two analyses. While this may raise important questions that need to be considered when generating and interpreting absolute HC_5 values derived from SSD curves, these questions are outside the scope of this work. For the purpose of this assessment, it was considered more important to ensure that consistent criteria were applied to the selection of endpoints for both GCH and LCH to allow a comparative analysis of the SSD curves to be conducted.

Fish

SSD curves for fish for GCH and LCH were constructed using the same methods as described above. The resulting SSD curves for GCH and LCH are shown in Fig. 2 (upper right panel). The fish HC_5 values for GCH and LCH differed by a factor of 1.72 (Table 4), which is consistent with GCH as a refined form of LCH.

Compared to the SSD for arthropods, the SSD for fish was shifted considerably to the right (less sensitive) for both GCH (Fig. 2, lower left panel) and LCH (Fig. 2, lower right panel). Regression parameters are shown in Table 4. The toxicity data clearly demonstrate that fish are less sensitive to LCH than are arthropods.

The fish HC_5 for LCH is 41 ng/L, nearly 40 times greater than the HC_5 for arthropods (Table 4) and 5 times

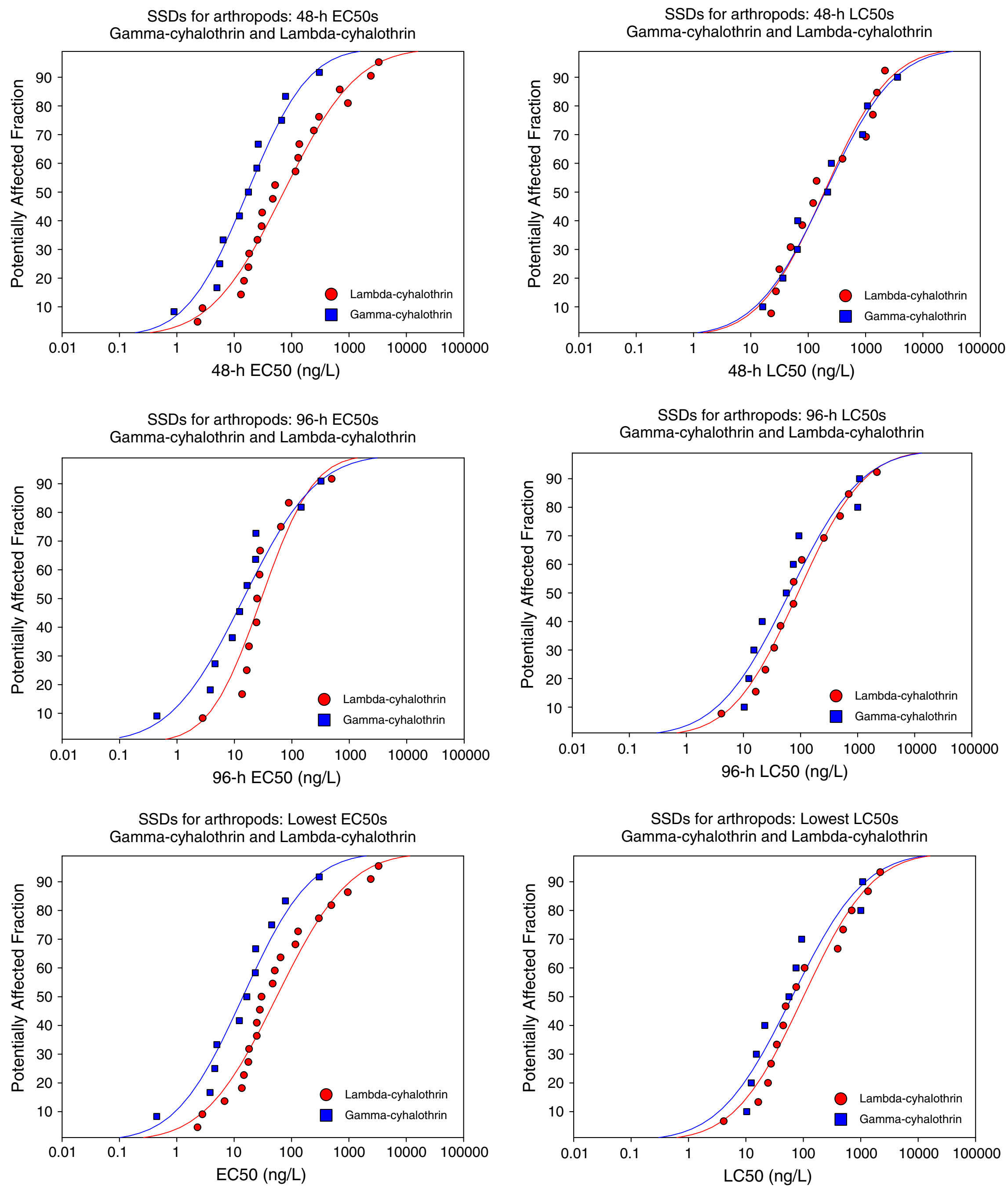

Fig. 1 Species sensitivity distributions for arthropods for gamma-cyhalothrin and lambda-cyhalothrin using different toxicity test endpoints

greater than the community-level no observed effect concentration ($NOEC_{community}$) for arthropods (see below). For GCH, the fish HC_5 is 24 ng/L, 50 times greater than the HC_5 for arthropods (Table 4) and 5 times greater than the arthropod $NOEC_{community}$. The finding of greater sensitivity of arthropods than fish to GCH and LCH based on HC_5 values is consistent with the conclusions of Maltby et al. (2005) for 16 insecticides, including LCH. For both LCH and GCH, a safe exposure level for arthropods (i.e., the $NOEC_{community}$) will also be safe for fish.

Field studies

The effects of LCH on aquatic communities under natural conditions have been extensively studied (Farmer et al. 1995; Hill et al. 1994a, b; Schroer et al. 2004; Roessink et al. 2005; Van Wijngaarden et al. 2006). Since LCH consists of ca. 50% GCH, and the toxicity profiles of LCH and GCH illustrate that GCH is the ecotoxicologically relevant component of LCH, information derived from field studies with LCH can also be directly applied to GCH.

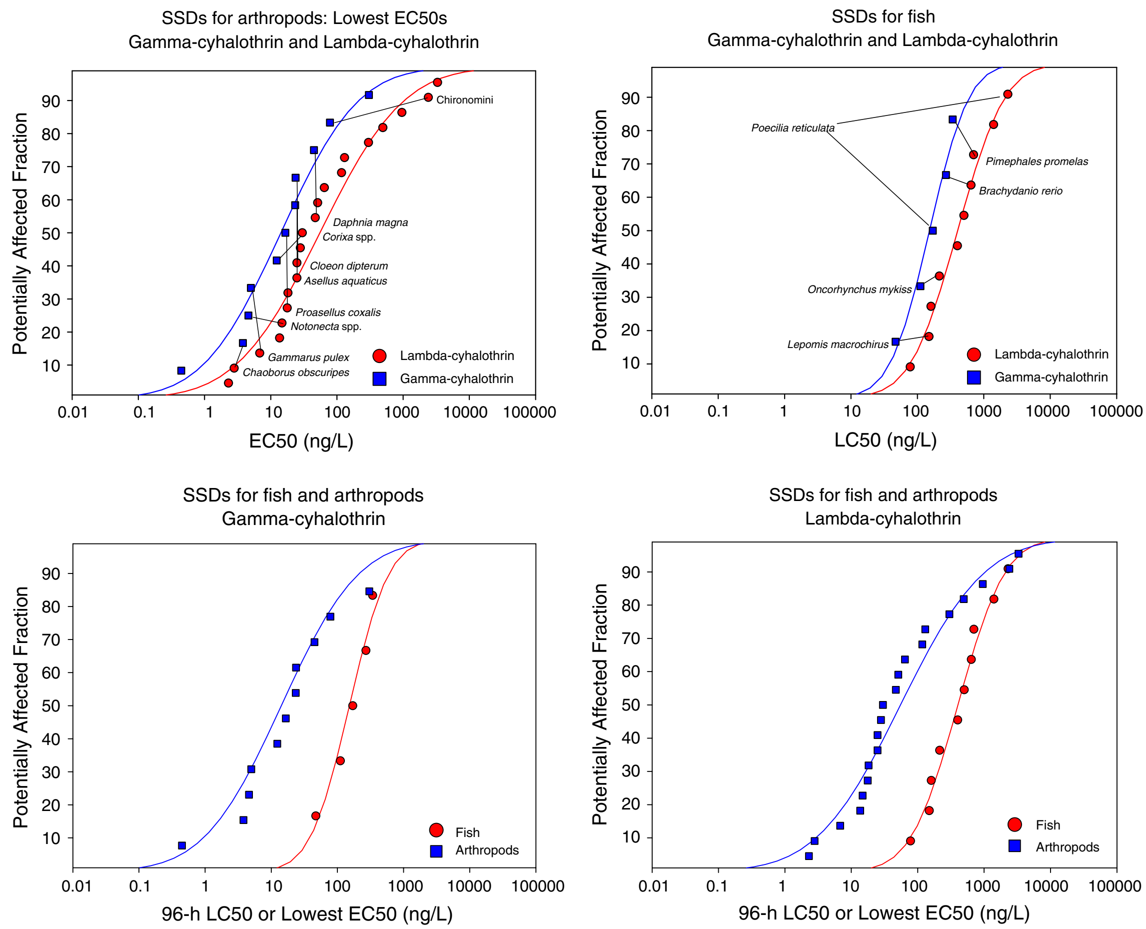

Fig. 2 Species sensitivity distributions for arthropods (lowest EC50s) and for fish (96-h LC50s) for gamma-cyhalothrin and lambda-cyhalothrin. In *upper panels*, EC50s and LC50s for individual species are connected by *straight lines*

Table 4 Parameters of species sensitivity distributions (based on logistic analysis) for gamma-cyhalothrin (GCH) and lambda-cyhalothrin (LCH)

	Arthropods (lowest EC50)		Fish (96-h LC50)	
	LCH	GCH	LCH	GCH
HC_5 (ng/L)	1.05	0.47	40.92	23.74
HC_5 prediction confidence interval	0.47–2.34	0.11–2.00	15.31–109.37	4.18–134.78
HC_{50} (ng/L)	48.51	14.61	400	157
HC_{50} prediction confidence interval	36.53–64.43	9.19–23.22	285–561	93–263
N	21	11	10	5
Slope	0.7676	0.8586	1.2912	1.5584
Intercept	−2.9797	−2.3024	−7.7367	−7.8797

To confirm this expectation, a microcosm study was conducted with GF-317, a capsule suspension formulation of GCH (van Wijngaarden et al. 2008). The study used macrophyte-dominated enclosures in the same set of experimental ditches as were used previously for LCH (Schroer et al. 2004; Roessink et al. 2005; van Wijngaarden et al. 2006). GF-317 was added three times at weekly intervals to give nominal GCH concentrations of 5, 10, 25, 50, and 100 ng/L in the water column. GCH dissipated rapidly from the water column, with only 40% remaining 24 h after application (similar to LCH, of which 30% remained after 24 h; Roessink et al. 2005). Zooplankton and macroinvertebrates were monitored closely; phytoplankton chlorophyll, macrophyte biomass, litter decomposition, and community metabolism were also measured.

The ecological responses observed in the enclosures treated with GCH were very similar to those observed in previous studies with LCH (Table 5). Based on multivariate principal response curve (PRC) analysis (Van den Brink and ter Braak 1998, 1999), the NOEC for the macroinvertebrate community was 5 ng/L in the GCH study, and less than 10 ng/L in the LCH study. PRC indicated consistent differences at 50 ng/L in the GCH study and at 100 ng/L in the LCH study. In both studies, the macroinvertebrate communities recovered from effects at the highest treatment level. PRC analysis indicated that the macroinvertebrates most affected by GCH were *Chaoborus* sp., *Caenis* sp., and *G. pulex*; the macroinvertebrates most affected by LCH were *G. pulex* and *C. obscuripes*. (*Caenis*, though not abundant in macrophyte-dominated enclosures in the LCH study, were among the most sensitive macroinvertebrates in plankton-dominated enclosures, consistent with the GCH enclosures). Based on univariate analysis in the GCH study, the abundance of the mayfly *Caenis* sp. was briefly reduced at 5 ng/L, and the midge subfamily

Table 5 Comparison of responses observed in enclosure studies with gamma-cyhalothrin (GCH) and lambda-cyhalothrin (LCH)

Subject	Measurement	GCH[a]	LCH[b]	Comparison
Fate	Dissipation from water	60% reduction in 24 h	70% reduction in 24 h	Similar
Macroinverts	PRC NOEC	5 ng/L	<10 ng/L	<2×
Macroinverts	PRC consistent reductions	50 ng/L	100 ng/L (inferred from Fig. 3 in Roessink et al. 2005)	2×
Macroinverts	PRC recovery	Within 59 day of final application at 100 ng/L	Within 28 day of final application at 100 ng/L	Similar
Macroinverts	PRC taxa most affected	*Chaoborus* sp.; *Caenis* sp.; *Gammarus pulex*	*Gammarus pulex*; *Chaoborus obscuripes*	Similar
Macroinverts	*Chaoborus* sp.	NOEC = 5 ng/L	NOEC = 10 ng/L	2×
Macroinverts	*Caenis* sp.	NOEC < 5 ng/L	Not reported	Comparison not possible
Macroinverts	Ceratopogonidae	NOEC = 10 ng/L	Not reported	Comparison not possible
Macroinverts	*Gammarus pulex*	NOEC = 25 ng/L	NOEC = 25 ng/L	Equal
Macroinverts	*Proasellus* spp.	NOEC = 25 ng/L	Not reported	Comparison not possible
Macroinverts	Orthocladiinae	NOEC < 5 ng/L	Not reported	Comparison not possible
Zooplankton	PRC NOEC	50 ng/L	25 ng/L	GCH less toxic than LCH
Zooplankton	PRC consistent reductions	50 ng/L	>250 ng/L	>5×
Zooplankton	PRC taxa most affected	*Cephalodella gibba*, *Colurella uncinata* (↑[c]); *Daphnia longispina*, *Ceriodaphnia quadrangula*	*Anureopsis fissa*; *Lecane gr. luna* (↑)	Similarities and differences
Zooplankton	Rotifers	*Cephalodella gibba*, *Colurella uncinata* NOEC = 50 ng/L (↑)	*Lecane lunaris* NOEC < 10 ng/L(↑)	Similar (neither toxic)
Zooplankton	Cladocerans	*Daphnia longispina* NOEC = 50 ng/L; *Ceriodaphnia quandangula* NOEC = 25 ng/L; total cladocera NOEC = 25 ng/L	Total cladocera NOEC < 10 ng/L(↑)	Opposite
Zooplankton	Copepods	Calanoida, Cyclopoida, nauplii NOEC = 100 ng/L	Nauplii NOEC = 25 ng/L	GCH less toxic than LCH
Phytoplankton	Chlorophyll	No effect	No effect	Equal
Macrophytes	Dry wt at termination	No effect	No effect	Equal
Decomposition	Litter bag	No effect	No effect	Equal
Metabolism	DO, pH	No effect	No effect	Equal
Community	$NOEC_{community}$	5 ng/L	10 ng/L	2×

[a] From Van Wijngaarden et al. (2008); macrophyte-dominated enclosures

[b] From Roessink et al. (2005); macrophyte-dominated enclosures

[c] (↑) Increase in abundance relative to controls

Orthocladiinae may also have been affected at 5 ng/L. The phantom midge *Chaoborus* was reduced at 10 ng/L, the midge family *Ceratopogonidae* at 25 ng/L, and the amphipod *G. pulex* and isopod *Proasellus* spp. at 50 ng/L. Where comparisons could be made, the effects of GCH on macroinvertebrate taxa were similar to or slightly greater than effects of similar concentrations of LCH. For example, the NOEC for *Chaoborus* was 5 ng/L for GCH and 10 ng/L for LCH; for *G. pulex*, the NOEC was 25 ng/L in both studies.

For the zooplankton community, PRC analysis indicated a NOEC of 50 ng/L for GCH, twice as great as in macrophyte-dominated LCH enclosures (Table 5). The cladocerans *Daphnia longispina* and *Ceriodaphnia quadrangula* showed the greatest negative responses to GCH treatment, while the rotifers *Cephalodella gibba* and *Colurella uncinata* showed the greatest positive responses. Univariate analysis of the GCH data indicated that the NOEC for *C. gibba* and *C. uncinata* (increased abundance) was 50 ng/L. NOECs (decreased abundance) were 50 ng/L for *D. longispina* and 25 ng/L for *C. quadrangula*. There were no effects on calanoid copepods, cyclopoid copepods, or copepod nauplii at 100 ng/L GCH, the highest treatment level. Some of these changes corresponded to observations in the LCH study, while others did not. For example, the rotifers *Lecane lunaris* and *Anureopsis fissa* increased in response to LCH, similar to the responses of other rotifer species to GCH. On the other hand, copepod nauplii decreased at 50 ng/L LCH, a response not observed with GCH, and cladocerans increased at 10 ng/L LCH but two cladoceran species decreased at 50 and 100 ng/L GCH. Overall, zooplankton were less affected than macroinvertebrates in both studies.

Neither GCH nor LCH affected phytoplankton chlorophyll concentrations, macrophyte biomass, or community metabolism. A small, transitory, non-dose-related reduction in litter decomposition rate was observed at 10 ng/L GCH, whereas LCH had no observed effect on litter decomposition.

When the results for each microcosm endpoint were expressed using the effect classes described by Brock et al. (2006), the Class 3 effects (short-term effects) were first noted at 10 ng/L for GCH (van Wijngaarden et al. 2008) and at 25 ng/L for LCH (Roessink et al. 2005). Long-term effects (Classes 4 and 5) were first noted at 25 ng/L for GCH and 100 ng/L for LCH. Taking all the ecological responses into account, the $NOEC_{community}$ was 5 ng/L for GCH, 10 ng/L for LCH. The difference between the $NOEC_{community}$ for GCH and LCH was consistent with the observed differences in single-species toxicity (Table 3) and in SSDs (Table 4). Like the laboratory toxicity results, the GCH enclosure results were consistent with GCH as a refined form of LCH.

Conclusions

Results for GCH toxicity to algae, invertebrates, and fish supported the inferred correspondence between the aquatic toxicity of GCH and LCH. Microcosm results with both substances were consistent with the single-species toxicity data. The data from toxicity tests and field studies indicated that GCH causes effects at approximately one-half the concentration at which LCH causes similar effects. These results are overwhelmingly consistent with GCH as a refined form of LCH. The difference in toxicity is small, within the range of variation generally expected from repeated studies with a single species and a single test substance in a single laboratory. Because GCH has approximately twice the biological activity of LCH, application rates needed for insect control are correspondingly lower, so the risks to aquatic species posed by LCH and GCH are essentially the same.

In light of the consistencies, and the extensive database available for GCH and LCH, there is a high degree of certainty about the likely effects of these pyrethroids on aquatic organisms. The high degree of certainty implies that the $NOEC_{community}$ based on the microcosm studies can be used to directly infer a "regulatory acceptable concentration" (RAC) for both GCH and LCH (Giddings et al. 2002).

References

Ali I, Gupta VK, Aboul-Enein HY (2003) Chirality: a challenge for the environmental scientists. Curr Sci 84(2):152–156

Brock TCM, Arts GHP, Maltby L, van den Brink P (2006) Aquatic risks of pesticides, ecological protection goals, and common aims in European Union legislation. Integr Environ Assess Manag 2:e20–e46. doi:10.1897/1551-3793(2006)2[e20:AROPEP]2.0.CO;2

Cai X, Liu W, Sheng G (2008) Enantioselective degradation and ecotoxicity of the chiral herbicide diclofop in three freshwater algal cultures. J Agric Food Chem 2008:2139–2146. doi:10.1021/jf0728855

Farmer D, Hill IR, Maund SJ (1995) A comparison of the fate and effects of two pyrethroid insecticides (lambda-cyhalothrin and cypermethrin) in pond mesocosms. Ecotoxicol 4:219–244. doi:10.1007/BF00116342

Giddings JM, Brock TCM, Heger W, Heimbach F, Maund SJ, Norman SM, Ratte HT, Schäfers C, Streloke M (2002) Community level aquatic system studies—interpretation and criteria (CLASSIC). Society of Environmental Toxicology and Chemistry, Pensacola

Henry KS, McClymont EL, Najar JR (2003a) XDE-225: an acute toxicity study with the amphipod, *Gammarus pseudolimnaeus*. Report No. 021059. Dow Chemical Company, Midland

Henry KS, McClymont EL, Najar JR (2003b) XDE-225: toxicity testing with various life stages of the amphipod *Gammarus pseudolimnaeus*. Study No. 021119. Dow Chemical Company, Midland

Hill IR, Runalls JK, Kennedy JH, Ekoniak P (1994a) Effects of lambda-cyhalothrin on aquatic organisms in large-scale mesocosms. In: Hill IR, Heimbach F, Leeuwangh P, Matthiessen P (eds) Freshwater field tests for hazard assessment of chemicals. Lewis Pubs, Boca Raton, pp 345–360

Hill IR, Runalls JK, Kennedy JH, Ekoniak P (1994b) Lambda-cyhalothrin: a mesocosm study of its effects on aquatic organisms. In: Graney RL, Kennedy JH, Rodgers JH (eds) Aquatic mesocosm studies in ecological risk assessment. Lewis Pub, Boca Raton, pp 403–468

Kirk HD, Giles MM, McClymont EL, McFadden LG (2000) XR-225: growth inhibition test with the freshwater green alga, *Selenastrum capricornutum* Printz. Study No. 001093. Dow Chemical Company, Midland

Kirk HD, Henry KS, Staley JL, McClymont EL, McFadden LG (2001) XDE-225: evaluation of the chronic toxicity with the daphnid, *Daphnia magna* Straus. Study No. 011051. Dow Chemical Company, Midland

Liu W, Gan J, Lee S, Werner I (2004) Isomer selectivity in aquatic toxicity and biodegradation of cypermethrin. J Agric Food Chem 52:6233–6238. doi:10.1021/jf0490910

Liu W, Gan J, Lee S, Werner I (2005a) Isomer selectivity in aquatic toxicity and biodegradation of bifenthrin and permethrin. Environ Toxicol Chem 24:1861–1866. doi:10.1897/04-457R.1

Liu W, Gan J, Schlenk D, Jury WA (2005b) Enantioselectivity in environmental safety of current chiral insecticides. Proc Natl Acad Sci USA 102:701–706. doi:10.1073/pnas.0408847102

Machado MW (2001a) XDE-225 and lambda-cyhalothrin: comparative toxicity to rainbow trout (*Oncorhynchus mykiss*) under flow-through conditions. Study No. 011056. Dow Chemical Company, Midland

Machado MW (2001b) XDE-225 and lambda-cyhalothrin: comparative toxicity to daphnids (*Daphnia magna*) under static-renewal conditions. Study No. 011050. Dow Chemical Company, Midland

Maltby L, Blake N, Brock TCM, van den Brink PJ (2005) Insecticide species sensitivity distributions: importance of test species selection and relevance to aquatic ecosystems. Environ Toxicol Chem 24:379–388. doi:10.1897/04-025R.1

Marino TA, Rick DL (2000a) XR-225: an acute toxicity study with the rainbow trout, *Oncorhynchus mykiss* Walbaum. Study No. 001073. Dow Chemical Company, Midland

Marino TA, Rick DL (2000b) XR-225: an acute toxicity study with the daphnia, *Daphnia magna* Strauss. Study No. 001075. Dow Chemical Company, Midland

Marino TA, Rick DL (2001a) XR-225: an acute toxicity study with the bluegill sunfish, *Lepomis macrochirus* Rafinesque. Study No. 001074(A). Dow Chemical Company, Midland

Marino TA, Rick DL (2001b) XR-225 and lambda-cyhalothrin: an acute toxicity comparison study with the bluegill sunfish, *Lepomis macrochirus* Rafinesque. Study No. 001074. Dow Chemical Company, Midland

Maund SJ, Hamer MJ, Warinton JS, Kedwards TJ (1998) Aquatic ecotoxicology of the pyrethroid insecticide lambda-cyhalothrin: considerations for higher-tier aquatic risk assessment. Pestic Sci 54:408–417. doi:10.1002/(SICI)1096-9063(199812)54:4<408::AID-PS843>3.0.CO;2-T

Posthuma L, Suter GW II, Traas TP (2002) Species sensitivity distributions in ecotoxicology. Lewis Publishers, Boca Raton

Roessink I, Arts GHP, Belgers JDM, Bransen F, Maund SJ, Brock TCM (2005) Effects of lambda-cyhalothrin in two ditch microcosm systems of different trophic status. Environ Toxicol Chem 24:1684–1696. doi:10.1897/04-130R.1

Schroer A, Belgers J, Brock T, Matser A, Maund S, van Den Brink P (2004) Comparison of laboratory single species and field population-level effects of the pyrethroid insecticide λ-cyhalothrin on freshwater invertebrates. Arch Environ Contam Toxicol 46:324–335. doi:10.1007/s00244-003-2315-3

Sewell IG, McKenzie J (2006a) Gamma-cyhalothrin: acute toxicity to fathead minnow (*Pimephales promelas*). SafePharm laboratories project No. 2119/0014. The Dow Chemical Company report 050607. April, 2006

Sewell IG, McKenzie J (2006b) Gamma-cyhalothrin: acute toxicity to guppy (*Poecilia reticulata*). SafePharm laboratories project No. 2119/0013. The Dow Chemical Company report 050606. April, 2006

Sewell IG, McKenzie J (2006c) Gamma-cyhalothrin: acute toxicity to zebra fish (*Brachydanio rerio*). SafePharm laboratories project No. 2119/0015. The Dow Chemical Company report 050608. April, 2006

U.S. EPA (United States Environmental Protection Agency) (2007) Office of pesticide programs (OPP) pesticide toxicity database. http://www.ipmcenters.org/Ecotox/index.cfm. Accessed Dec 2007

Van den Brink PJ, ter Braak CJF (1998) Multivariate analysis of stress in experimental ecosystems by principal response curves and multivariate analysis. Aquat Ecol 32:163–178. doi:10.1023/A:1009944004756

Van den Brink PJ, ter Braak CJF (1999) Principal response curves: analysis of time-dependent multivariate responses of a biological community to stress. Environ Toxicol Chem 18:138–148. doi:10.1897/1551-5028(1999)018<0138:PRCAOT>2.3.CO;2

Van Wijngaarden RPA, Barber I, Brock TCM (2008) Effects of the pyrethroid insecticide gamma-cyhalothrin on aquatic invertebrates in laboratory and outdoor microcosm tests. Ecotoxicology. doi:10.1007/s10646-008-0274-1

Van Wijngaarden RPA, Brock TCM, van den Brink PJ, Gylstra R, Maund SJ (2006) Ecological effects of spring and late summer applications of lambda-cyhalothrin on freshwater microcosms. Arch Environ Contam Toxicol 50:220–239. doi:10.1007/s00244-004-0249-z

Xu C, Wang J, Liu W, Sheng G, Tu Y, Ma Y (2008) Separation and aquatic toxicity of enantiomers of the pyrethroid insecticide lambda-cyhalothrin. Environ Toxicol Chem 27:174–181. doi:10.1897/07-134.1