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Lambda-Cyhalothrin
Human Health and Ecological Risk Assessment
Final Report

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ACRONYMS, ABBREVIATIONS, AND SYMBOLS

ACGIH	American Conference of Governmental Industrial Hygienists
AChE	acetylcholinesterase
AEL	adverse-effect level
a.i.	active ingredient
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	bioconcentration factor
bw	body weight
calc	calculated value
CBI	confidential business information
ChE	cholinesterase
CI	confidence interval
cm	centimeter
CNS	central nervous system
DAT	days after treatment
DER	data evaluation record
DOC	dissolved organic carbon
d.f.	degrees of freedom
EC	emulsifiable concentrate (in reference to a formulation)
EC _x	concentration causing X% inhibition of a process
EC ₂₅	concentration causing 25% inhibition of a process
EC ₅₀	concentration causing 50% inhibition of a process
EHE	2-ethylhexyl ester
EFED	Environmental Fate and Effects Division (U.S. EPA/OPP)
ExToxNet	Extension Toxicology Network
F	female
FH	Forest Health
FIFRA	Federal Insecticide, Fungicide and Rodenticide Act
FQPA	Food Quality Protection Act
g	gram
GLP	Good Laboratory Practices
ha	hectare
HED	Health Effects Division (U.S. EPA/OPP)
HQ	hazard quotient
IARC	International Agency for Research on Cancer
IREED	Interim Reregistration Eligibility Decision
IRIS	Integrated Risk Information System
k _a	absorption coefficient
k _e	elimination coefficient
kg	kilogram
K _{o/c}	organic carbon partition coefficient
K _{o/w}	octanol-water partition coefficient
K _p	skin permeability coefficient
L	liter
lb	pound
LC ₅₀	lethal concentration, 50% kill
LD ₅₀	lethal dose, 50% kill

ACRONYMS, ABBREVIATIONS, AND SYMBOLS *(continued)*

LOAEL	lowest-observed-adverse-effect level
LOC	level of concern
m	meter
M	male
mg	milligram
mg/kg/day	milligrams of agent per kilogram of body weight per day
mL	milliliter
mM	millimole
mPa	millipascal, (0.001 Pa)
MOS	margin of safety
MRID	Master Record Identification Number
MSDS	material safety data sheet
MSMA	monosodium methanearsonate
MW	molecular weight
NAWQA	USGS National Water Quality Assessment
NCI	National Cancer Institute
NCOD	National Drinking Water Contaminant Occurrence Database
NIOSH	National Institute for Occupational Safety and Health
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
NOS	not otherwise specified
NRC	National Research Council
NTP	National Toxicology Program
OM	organic matter
OPP	Office of Pesticide Programs
OPPTS	Office of Pesticide Planning and Toxic Substances
OSHA	Occupational Safety and Health Administration
Pa	Pascal
PBPK	physiologically-based kinetic
ppm	parts per million
RBC	red blood cells
RED	re-registration eligibility decision
RfD	reference dose
SERA	Syracuse Environmental Research Associates
TEP	typical end-use product
T.G.I.A.	Technical grade active ingredient
TIPA	Triisopropanolamine
TRED	Tolerance Reassessment Eligibility Decision
UF	uncertainty factor
U.S.	United States
USDA	U.S. Department of Agriculture
U.S. EPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
WHO	World Health Organization

COMMON UNIT CONVERSIONS AND ABBREVIATIONS

To convert ...	Into ...	Multiply by ...
acres	hectares (ha)	0.4047
acres	square meters (m ²)	4,047
atmospheres	millimeters of mercury	760
centigrade	Fahrenheit	1.8°C+32
centimeters	inches	0.3937
cubic meters (m ³)	liters (L)	1,000
Fahrenheit	centigrade	0.556°F-17.8
feet per second (ft/sec)	miles/hour (mi/hr)	0.6818
gallons (gal)	liters (L)	3.785
gallons per acre (gal/acre)	liters per hectare (L/ha)	9.34
grams (g)	ounces, (oz)	0.03527
grams (g)	pounds, (oz)	0.002205
hectares (ha)	acres	2.471
inches (in)	centimeters (cm)	2.540
kilograms (kg)	ounces, (oz)	35.274
kilograms (kg)	pounds, (lb)	2.2046
kilograms per hectare (kg/ha)	pounds per acre (lb/acre)	0.892
kilometers (km)	miles (mi)	0.6214
liters (L)	cubic centimeters (cm ³)	1,000
liters (L)	gallons (gal)	0.2642
liters (L)	ounces, fluid (oz)	33.814
miles (mi)	kilometers (km)	1.609
miles per hour (mi/hr)	cm/sec	44.70
milligrams (mg)	ounces (oz)	0.000035
meters (m)	feet	3.281
ounces (oz)	grams (g)	28.3495
ounces per acre (oz/acre)	grams per hectare (g/ha)	70.1
ounces per acre (oz/acre)	kilograms per hectare (kg/ha)	0.0701
ounces fluid	cubic centimeters (cm ³)	29.5735
pounds (lb)	grams (g)	453.6
pounds (lb)	kilograms (kg)	0.4536
pounds per acre (lb/acre)	kilograms per hectare (kg/ha)	1.121
pounds per acre (lb/acre)	mg/square meter (mg/m ²)	112.1
pounds per acre (lb/acre)	µg/square centimeter (µg/cm ²)	11.21
pounds per gallon (lb/gal)	grams per liter (g/L)	119.8
square centimeters (cm ²)	square inches (in ²)	0.155
square centimeters (cm ²)	square meters (m ²)	0.0001
square meters (m ²)	square centimeters (cm ²)	10,000
yards	meters	0.9144

Note: All references to pounds and ounces refer to avoirdupois weights unless otherwise specified.

CONVERSION OF SCIENTIFIC NOTATION

Scientific Notation	Decimal Equivalent	Verbal Expression
$1 \cdot 10^{-10}$	0.0000000001	One in ten billion
$1 \cdot 10^{-9}$	0.000000001	One in one billion
$1 \cdot 10^{-8}$	0.00000001	One in one hundred million
$1 \cdot 10^{-7}$	0.0000001	One in ten million
$1 \cdot 10^{-6}$	0.000001	One in one million
$1 \cdot 10^{-5}$	0.00001	One in one hundred thousand
$1 \cdot 10^{-4}$	0.0001	One in ten thousand
$1 \cdot 10^{-3}$	0.001	One in one thousand
$1 \cdot 10^{-2}$	0.01	One in one hundred
$1 \cdot 10^{-1}$	0.1	One in ten
$1 \cdot 10^0$	1	One
$1 \cdot 10^1$	10	Ten
$1 \cdot 10^2$	100	One hundred
$1 \cdot 10^3$	1,000	One thousand
$1 \cdot 10^4$	10,000	Ten thousand
$1 \cdot 10^5$	100,000	One hundred thousand
$1 \cdot 10^6$	1,000,000	One million
$1 \cdot 10^7$	10,000,000	Ten million
$1 \cdot 10^8$	100,000,000	One hundred million
$1 \cdot 10^9$	1,000,000,000	One billion
$1 \cdot 10^{10}$	10,000,000,000	Ten billion

EXECUTIVE SUMMARY

This risk assessment focuses on the potential use of an insecticide, lambda-cyhalothrin, at two sites in California, the Chico Genetic Resources and Conservation Center located in the Mendocino National Forest and the Foresthill Genetics Center located in the Tahoe National Forest. For brevity, these sites are designated in this risk assessment as the Chico site and Foresthill site, respectively. The Forest Service is considering the use of lambda-cyhalothrin as an alternative to or in addition to the use of esfenvalerate to control coneworm (*Dioryctria* spp.), seed bugs (*Leptoglossus* spp.), and cone beetles (*Conophthorus* spp.) at these sites. Both lambda-cyhalothrin and esfenvalerate are pyrethroids. If these two insecticides are used at the same site over the course of 1 year, it would be prudent to regard risks associated with esfenvalerate and lambda-cyhalothrin as additive.

The exposures specifically considered in this risk assessment are based on six single applications of 0.08 lb a.i./acre with a 2-week interval between applications. Since lambda-cyhalothrin is not currently used at the two sites in California for which this risk assessment is developed, the Forest Service may consider using somewhat lower or higher application rates (up to 0.16 lb a.i./acre) resulting in a cumulative annual application rate of 0.5 lb a.i./acre. Although the different rates would have an impact on the specific HQs, the qualitative risk characterization would not change substantially over any plausible range of application rates.

Risks to workers appear to be low. Under the application methods and worker protection measures considered by the Forest Service, there is no basis for asserting that workers are likely to be at risk in the normal application of lambda-cyhalothrin at the Chico or Foresthill sites. As with almost all insecticide applications, accidental exposures are a concern. Nonetheless, the risks of systemic toxicity are probably low, so long as prudent worker protection measures are implemented effectively.

For members of the general public, the quantitative risk characterizations are different for the Chico and Foresthill sites; however, these differences may reflect the fact that the Chico site is much better characterized, relative to the Foresthill site, at least in terms of the plausible exposures for members of the general public. At the Chico site, the most plausible exposure scenario involves the consumption of contaminated blackberries from bushes growing along the banks of Comanche Creek. The HQs for these exposure scenarios are below the level of concern by a factor of at least 10. The only other plausible non-accidental exposure scenario involves the consumption of fish by subsistence populations. The upper bound HQ for this scenario is below the level of concern by a factor of 3. Much higher HQs are derived for the Foresthill site because very conservative default exposure assumptions are used, in the absence of specific information justifying the use of other exposure assumptions. For the Foresthill site, the HQs for the consumption of contaminated vegetation and fruit exceed the level of concern by factors of 4-35. Because the Foresthill site is in a relatively remote location, the risk characterization for this site may reflect potential rather than plausible risk. Accidental exposure scenarios for both sites result in HQs that substantially exceed the level of concern, which is typical of risk characterizations for many pesticides covered by Forest Service risk assessments. In the event of

1 major accidental spills or other accidental events, remedial actions to reduce and limit exposures
2 to members of the general public would be appropriate.

3
4 Lambda-cyhalothrin is an effective insecticide. Within the treated area, terrestrial insects will be
5 adversely affected (and probably killed) in any effective application of lambda-cyhalothrin.
6 Insects not present at the application site will be at much lower risk. Lambda-cyhalothrin is also
7 highly toxic to some fish and aquatic arthropods. Peak concentrations of lambda-cyhalothrin are
8 likely to cause substantial mortality in sensitive species of fish and aquatic arthropods. For the
9 Foresthill site, potential effects on fish are not a practical concern because fish do not inhabit the
10 creek at this site. Longer-term concentrations of lambda-cyhalothrin in surface water could also
11 adversely affect sensitive species of aquatic arthropods; however, these concentrations are not
12 likely to have an impact on even sensitive species of fish. The relatively high HQ values for
13 sensitive species of fish and aquatic arthropods raise concern for downstream contamination.
14 The risks associated with downstream contamination are not quantified in this risk assessment
15 due to the lack of sufficient information on the flow velocities and flow volumes of the creeks
16 which might be affected.

17
18 Plausible risks to mammals, soil microorganisms, terrestrial plants, or aquatic plants cannot be
19 identified from the available information on lambda-cyhalothrin. Furthermore, risks to birds and
20 non-arthropod aquatic invertebrates are not likely to be substantial. The only concern for non-
21 arthropod aquatic invertebrates involves larval stage mollusks or adult mollusks without shells,
22 for which data are not available. It is not clear that larval stage mollusks and adult mollusks
23 without shells would be as tolerant as adult stage mollusks with shells to the effects of lambda-
24 cyhalothrin exposure. Risks to amphibians cannot be characterized directly; however, it is
25 reasonable to speculate that the range of sensitivity among amphibians may be similar to that of
26 fish.

1. INTRODUCTION

This document is an abbreviated risk assessment of the human health and ecological effects associated with the use of lambda-cyhalothrin in USDA Forest Service programs. Lambda-cyhalothrin is an insecticide which is being considered to control the coneworm (*Dioryctria* spp.), seed bugs (*Leptoglossus* spp.), and cone beetles (*Conophthorus* spp.) at two sites in California. Currently, esfenvalerate is used to control these pests. This risk assessment focuses on the use of lambda-cyhalothrin at these two sites only. In the event that the Forest Service wishes to consider using lambda-cyhalothrin at a wider range of locations, the risk assessment covers the more general use of the insecticide in other regions of the United States.

The information covered in this risk assessment is based on a standard search of TOXLINE as well as available reviews and assessments by the U.S. EPA (U.S. EPA/OPP 1988a; U.S. EPA/OPP 2002a; U.S. EPA/OPP 2004a; U.S. EPA/ORD 1988), the World Health Organization (WHO 1990a,b), and the Agency for Toxic Substances and Disease Registry (ATSDR 2003), and the California Department of Pesticide Regulation (<http://www.cdpr.ca.gov/docs/registration/reevaluation/chemicals/pyrethroids.htm>). An additional source of information included all cleared reviews that are available from the U.S. EPA/OPP (<http://www.epa.gov/pesticides/foia/reviews/128897/index.htm>) as well as a search of the E-Docket (<http://www.regulations.gov>) for entries related to lambda-cyhalothrin. A total of 107 cleared reviews relevant to this Forest Service risk assessment were downloaded and are included in the reference list (Section 5). The only documents from the E-Docket search to be reviewed in detail are those that appeared to be directly related to quantitative assessments of risks—e.g., pesticide tolerances. U.S. EPA/OPP is in the process of updating the risk assessments for pyrethroids (<http://www.epa.gov/oppsrrd1/reevaluation/pyrethroids-pyrethrins.html#epa>). At the time this Forest Service risk assessment was prepared, however, REDs or related documents (i.e., science chapters) on or covering lambda-cyhalothrin were not identified.

The current risk assessment is abbreviated in the interest of economy because of the limited uses of lambda-cyhalothrin under consideration by the Forest Service (i.e., applications at only two sites). More specifically, efficacy studies were not obtained or reviewed unless the information from the literature search suggested that the efficacy study might contain information on effects to nontarget organisms or sublethal toxicity (e.g., Abro et al. 1987; Li and Harmsen 1993). Emphasis is placed on studies conducted in the United States, particularly studies conducted in California. Studies conducted outside of the United States were not reviewed unless they appeared to be relevant in terms of nontarget effects or the development of resistance in target species. Furthermore, the information and discussion presented in the current Forest Service risk assessment is less detailed than that in standard Forest Service risk assessments—e.g., fewer and less detailed appendices are included, and some topics relevant to the hazard identification place greater reliance on reviews by U.S. EPA and ATSDR than is typical in most Forest Service risk assessments.

1 The Forest Service may elect to update and/or expand the current risk assessment and
2 welcomes input from the general public on the selection of studies included in the risk
3 assessment. This input is helpful, however, only if recommendations for including
4 additional studies specify why and/or how the new or not previously included
5 information would be likely to alter the conclusions reached in the risk assessments.
6

7 This document includes an introduction, program description, risk assessment for human
8 health effects, and risk assessment for ecological effects or effects on wildlife species.
9 Each of the two risk assessment chapters has four major sections, including an
10 identification of the hazards associated with lambda-cyhalothrin and its commercial
11 formulation, an assessment of potential exposure to the products, an assessment of the
12 dose-response relationships, and a characterization of the risks associated with plausible
13 levels of exposure. These major sections represent the basic steps recommended by the
14 National Research Council of the National Academy of Sciences (NRC 1983) for
15 conducting and organizing risk assessments.
16

17 Although this is a technical support document and addresses some specialized technical
18 areas, an effort was made to ensure that the document can be understood by individuals
19 who do not have specialized training in the chemical and biological sciences. Certain
20 technical concepts, methods, and terms common to all parts of the risk assessment are
21 described in plain language in a separate document (SERA 2007a).
22

23 As with all Forest Service risk assessments, risks are typically expressed as a central
24 estimate and a range, which is sometimes quite large. Because of the need to encompass
25 many different types of exposure as well as the need to express the uncertainties in the
26 assessment, this risk assessment involves numerous calculations. Relatively simple
27 calculations are included in the body of the document. For more cumbersome
28 calculations, two EXCEL workbooks, consisting of sets of EXCEL worksheets, are
29 included as attachments to the risk assessment: a workbook for the Chico site
30 (Attachment I) and a workbook for the Foresthill site (Attachment 2). The worksheets
31 provide the detail for the estimates cited in the body of this document. Documentation on
32 the use of the EXCEL worksheets is provided in SERA (2009a).

1

2. PROGRAM DESCRIPTION

2

2.1. OVERVIEW

3 Lambda-cyhalothrin is an insecticide that is being considered by the Forest Service as an
4 alternative to or in addition to esfenvalerate for the control of control of coneworm
5 (*Dioryctria* spp.), seed bugs (*Leptoglossus* spp.), and cone beetles (*Conophthorus* spp.).
6 At this time, the Forest Service is considering using lambda-cyhalothrin at only two sites
7 located in California—i.e., the Chico Genetic Resources and Conservation Center located
8 in the Mendocino National Forest and the Foresthill Genetics Center located in the Tahoe
9 National Forest. For brevity, these sites are designated in this risk assessment as the
10 Chico Site and Foresthill site, respectively.

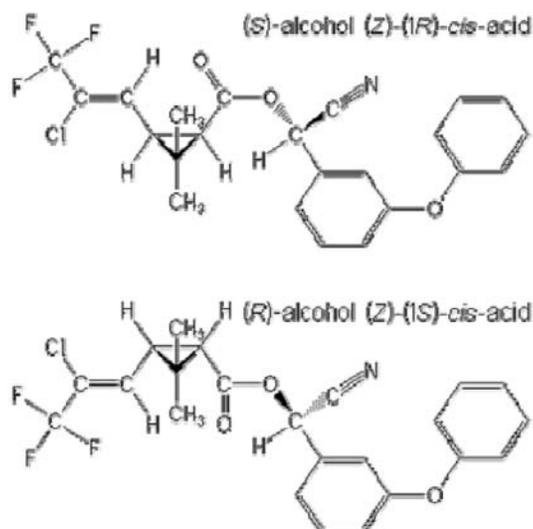
11
12 Lambda-cyhalothrin is produced by Syngenta, LG Life Sciences, Ltd, United Phosphorus
13 Inc., and Helm Agro US, Inc., and more than 170 formulations are available. The current
14 risk assessment explicitly considers formulations identified by the Forest Service as
15 candidates for use. The information included in the risk assessment, however, should
16 support an analysis of any formulation registered by the EPA for the pest species of
17 concern. Only two application methods are currently being considered by the Forest
18 Service, low volume ground-based broadcast orchard sprayer (at the Chico site) and high
19 volume individual-tree spray (Foresthill site).

20
21 For low volume ground applications, the maximum labeled rate for a single application is
22 0.16 lb a.i./acre with a maximum cumulative annual application rate of 0.5 lb a.i./acre.
23 Thus, up to three applications at 0.16 lb a.i./acre could be made each year. The Forest
24 Service does anticipate multiple applications with an application interval of 2-4 weeks.
25 For high volume applications to individual trees, application rates in units of lb a.i./acre
26 are not given on the product labels. The mixing and application instructions on the
27 labels, however, lead to application rates of 0.002-0.004 lb a.i./tree. Nonetheless, the
28 maximum annual application rate of 0.5 lb a.i./acre is applicable to tree applications as
29 well as airblast applications.

30

2.2. CHEMICAL DESCRIPTION AND COMMERCIAL FORMULATIONS

31 Lambda-cyhalothrin is the common name for a 1:1 mixture of two enantiomers (i.e.,
32 stereoisomers that are nonsuperimposable mirror images) of a phenoxybenzyl
33 halogenated cyclopropane-carboxylate:



1
2
3 The more formal nomenclature and the physical and chemical properties of lambda-
4 cyhalothrin are summarized in Table 1. A closely related pesticide, gamma-cyhalothrin,
5 consists only of the (S)-alcohol isomer (Dow Chemical Company 2008; Wood 2009a,b;
6 U.S. EPA-OPP 2007a). Notably, the generic term, *cyhalothrin*, with neither the lambda-
7 or gamma- designation, is a mixture of four isomers—i.e., mixtures of the + (cis) or –
8 (cis) acid and the (S)- or (R)- alcohols (WHO 1990a).

9
10 Lambda-cyhalothrin is a member of a class of insecticides known as pyrethroids.
11 Pyrethroids are synthetic or man-made insecticides designed to mimic a class of naturally
12 occurring pesticides known as pyrethrins. As discussed further in Section 3.2
13 (Mechanisms of Action), both pyrethrins and pyrethroids are neurotoxins that interfere
14 with the normal function of sodium channels in nerve cells (ATSDR 2003).

15
16 Technical grade lambda-cyhalothrin is produced by Syngenta (U.S. EPA/OPP 2004b),
17 LG Life Sciences, Ltd (U.S. EPA/OPP 2006a), United Phosphorus Inc. (U.S. EPA/OPP
18 2005a), and Helm Agro US, Inc (U.S. EPA/OPP 2009a). More than 170 formulations of
19 lambda-cyhalothrin are currently registered in the United States (PAN Pesticides
20 Database at <http://www.pesticideinfo.org>). A selected but not necessarily inclusive list of
21 formulations under consideration by the Forest Service (Bakke 2009) is provided in Table
22 2. All of these formulations are labeled for the control of coneworm and seed bug
23 species as well as for many other insects.

24
25 Except for Kaiso 24WG, all of the nine formulations listed in Table 2 are liquid
26 formulations, and five of the eight liquid formulations consist of an 11.4% solution of
27 lambda-cyhalothrin which contains 1 lb a.i./gallon. The inerts in the liquid formulations
28 of lambda-cyhalothrin consist generally of petroleum distillates (Table 3). As indicated
29 in Table 1, lambda-cyhalothrin is a highly lipophilic chemical—i.e., it has a very high
30 octanol-water partition coefficient (10,000,000) and very low water solubility (0.005
31 mg/L). Consequently, liquid formulations of lambda-cyhalothrin all contain petroleum
32 solvents. Generally, all pyrethrins and pyrethroids are formulated using petroleum
33 distillates and emulsifiers (ATSDR 2003, p. 153 ff).

1
2 As summarized in Table 3, the formulations designated as Grizzly X, Lambda-T, and
3 Taiga-Z contain the same specifications: 11.4% a.i., $\leq 1.4\%$ naphthalene, and unspecified
4 amounts of propylene glycol and an unspecified petroleum solvent. The formulation
5 specification for Warrior is quite similar to these formulations differing only slightly in
6 the specification of the amount of naphthalene—i.e., $< 1.5\%$ in Warrior vs $\leq 1.4\%$ for
7 Grizzly X, Lambda-T, and Taiga-Z. Warrior II does not specify the proportion of any
8 inerts, indicating only that the inerts are a petroleum solvent and titanium dioxide. The
9 MSDS for these formulations designate unspecified inerts as ranging from about 77 to
10 88% of the formulations.

11
12 U.S. EPA/OPPTS (2003, p. 5-2) encourages but does not require expanded inert
13 statements on product labels which specifically identify the inert ingredients in the
14 product label. Relatively detailed inert statements are presented only on the product
15 labels for the Lambda-Cy and Silencer formulations. Lambda-Cy specifies only one
16 designated inert, Solvesso 200. Nonetheless, relatively detailed information on this inert
17 is available. Solvesso 200 is a solution of naphtha, a heavy aromatic petroleum distillate
18 that contains naphthalene ($\leq 14\%$) as well as 1-methylnaphthalene ($\leq 12.5\%$) and 2-
19 methylnaphthalene ($\leq 26.5\%$) (Exxon Mobil 2007). Only 10.6% of the inerts in Lambda-
20 Cy are unspecified. The inerts in the Silencer formulation are more clearly specified—
21 i.e., 74.8% of an aromatic solvent with a designated CAS number and 7.84%
22 naphthalene. Only 1.06% of the inerts in Silencer are unspecified.

23
24 While the liquid formulations of lambda-cyhalothrin differ in the way in which the inerts
25 are specified, most of these liquid formulations appear to be generally similar to liquid
26 formulations of other pyrethroids—i.e., emulsifiers and petroleum distillates (ATSDR
27 2003). The potential impact of these inerts on the consideration of lambda-cyhalothrin
28 formulations is discussed further in Section 3.1.14 (Inerts and Adjuvants).

29
30 Lambdastar 1 CS is somewhat unusual in that no inerts are specified on either the product
31 label or the MSDS. The MSDS states that: *Ingredients not precisely identified are*
32 *proprietary or non-hazardous*. U.S. EPA requires that products containing $> 10\%$
33 petroleum distillates, xylene, or other xylene range aromatics contain a statement
34 identifying these inerts at least qualitatively on the product label (U.S. EPA/OPPTS 2003,
35 p. 5-6, 5-11) as well as a cautionary note to physicians concerning the potential for
36 vomiting to cause aspiration pneumonia. Although the MSDS and product label contain
37 cautionary language on vomiting, they do not specifically indicate that petroleum
38 distillates are included in the formulation. The product label for this formulation was
39 reviewed by U.S. EPA/OPP (unsigned letter dated April 9, 2009) and no comments
40 addressing inerts are made in the EPA review. Thus, if Lambdastar 1 CS contains
41 petroleum distillates, the concentration may be $< 10\%$.

42
43 The one granular formulation of lambda-cyhalothrin, Kaiso 24WG, contains only one
44 designated inert, N-methyl pyrrolidone. N-methyl pyrrolidone is used in many pesticide
45 formulations as a solvent, and this agent is also discussed in Section 3.1.14 (Inerts and
46 Adjuvants). As detailed further in Section 2.4 (Mixing and Application Rates), the

1 labeled mixing directions for Kaiso 24WG do not recommend the use of petroleum based
2 solvents. Thus, applications of Kaiso 24WG differ from applications of other lambda-
3 cyhalothrin formulations in that the potential effects of petroleum based solvents are not a
4 consideration.

5
6 All formulations of lambda-cyhalothrin are classified as restricted use pesticides—i.e.,
7 they may be applied only by certified applicators or individuals under the direct
8 supervision of certified pesticide applicators. The rationale for the restricted use
9 classification specified on the product labels involves the toxicity of lambda-cyhalothrin
10 to fish and other aquatic organisms, as discussed further in Section 4.1.3.

11
12 The literature on lambda-cyhalothrin contains many publications that refer to Karate
13 formulations. Two Karate formulations are available from Syngenta, a 13.1% liquid
14 (Karate) and a 22.8% liquid (Karate with Zeon Technology). These formulations are not
15 included in Table 2 because they are not registered in California; however, their use in
16 other Forest Service regions, are encompassed in this risk assessment.

17 **2.3. APPLICATION METHODS**

18 The Forest Service is considering only two foliar application methods for lambda-
19 cyhalothrin. At the Chico site, the application would be via an orchard air blast sprayer,
20 towed behind a tractor with an enclosed cab. At the Foresthill Site, applications would be
21 made to individual trees using high-pressure nozzles. In these applications, workers will
22 wear waterproof Tyvek overalls with a built-in hood, full face shield, protective glasses,
23 rubber boots, and chemical gloves taped into the coveralls. The impact of the use of
24 personal protective equipment is considered further in the exposure assessment for
25 workers.

26
27 All applications would be made at wind speeds of less than 15 miles per hour. Typically,
28 two workers would be involved in an application. One individual would prepare the tank
29 mix and monitor safety procedures. The other individual would apply the pesticide. The
30 applicator would wear coveralls, rubber boots, and a hard hat. The use of personal
31 protective equipment is discussed further in Section 3.2.2 (Exposure Assessment for
32 Workers).

33
34 Typically, applications could be made from April through August, up to twice a month,
35 depending on the tree species (sugar pine, ponderosa pine, Douglas-fir, white fir).
36 Timing for Douglas-fir would be earlier, perhaps starting as early as April, and probably
37 in May for white fir or sugar pine. Final treatments would be in July (for August
38 collection of Douglas-fir seeds) or August (for September collection of at least sugar pine
39 seed).

1 **2.4. MIXING AND APPLICATION RATES**

2 **2.3.1. Foliar Air Blast Applications (Chico Site)**

3 All of the labels for the lambda-cyhalothrin formulations listed in Table 2 recommend
4 application rates about 0.16 lb a.i./acre for the control of coneworms or seed bugs. As
5 summarized in Table 3, most of the liquid formulations contain 1 lb a.i./gallon. For these
6 formulations, label directions for low volume ground-based applications specify that
7 20 fl. oz should be added to each 100 gallons of water and that 100 gallons of the finished
8 solution should be applied to each acre. Thus, the application rate is about 0.16 lb
9 a.i./acre [$20 \text{ oz} \div (128 \text{ oz/gal}) \times (1 \text{ lb a.i./gal})/\text{acre} = 0.15625 \text{ lb a.i./acre}$]. The Warrior II
10 formulation contains 2.08 lb a.i./gallon and specifies a low volume ground spray rate of
11 10 fl. oz/100 gallons with an application rate of 100 gallons of finished spray per acre.
12 This also corresponds to an application rate of about 0.16 lb a.i./acre [$10 \text{ oz} \div (128$
13 $\text{oz/gal}) \times (2.08 \text{ lb a.i./gal})/\text{acre} = 0.1625 \text{ lb a.i./acre}$], although the precise application rate
14 is somewhat higher than that for the 1 lb a.i./gallon formulations. The product for the
15 Kaiso 24WG granular formulation (24% a.i.) indicates that 10.4 oz (avoirdupois) should
16 be added to 100 gallons of water and that 100 gallons of the finished solution should be
17 applied per acre. This corresponds to an application rate of 0.156 lb a.i./acre [10.4
18 $\text{oz/acre} \times 0.24 \times 1 \text{ lb}/16 \text{ oz} = 0.156 \text{ lb a.i./acre}$].

19
20 The product labels for all formulations of lambda-cyhalothrin indicate that the maximum
21 annual application rate for the control of coneworm and seed bug species is 0.5 lb
22 a.i./acre. Thus, at the recommended broadcast application rate of 0.16 lb a.i./acre,
23 lambda-cyhalothrin could be applied up to three times per year. The product labels do
24 not recommend a specific application interval for the control of coneworm and seed bug
25 species.

26
27 Applications at the Chico site are made between 5:00 and 10:00 PM and involve five
28 operators, each using 500 gallon tanks towed by a tractor with an enclosed, air-
29 conditioned cabin. To further reduce worker exposure, the door in the tractor cabin is
30 covered with heavy plastic stripping, ceiling to floor (Bakke 2009c).

31 **2.3.2. Applications to Individual Trees (Forest Hill Site)**

32 In applications to single trees, all of the formulations listed in Table 2 recommend
33 application rates of about 0.002-0.004 lb a.i./tree. The liquid formulations containing
34 1 lb a.i./acre indicate that 5.12 fl. oz. of formulation should be used for each 100 gallons
35 of finished solution and that 5-10 gallons should be applied per tree, which corresponds
36 to application rates of precisely 0.002-0.004 lb a.i./tree [$(1 \text{ lb/gallon formulation} \times 5.12$
37 $\text{oz formulation} \times 1 \text{ gallon}/128 \text{ oz} \div 100 \text{ gallons}) \times 5\text{-}10 \text{ gallons/tree}$]. The product label
38 for the Warrior II formulation (2.08 lb a.i./gallon) specifies that 2.56 fl oz should be used
39 per 100 gallons of water and that 5-10 gallons should be applied per tree. Because the
40 Warrior II formulation contains somewhat more than twice the amount of a.i., these
41 directions lead to modestly and insignificantly higher application rates of 0.00208-
42 0.00416 lb a.i./tree [$(2.08 \text{ lb/gallon formulation} \times 2.56 \text{ oz formulation} \times 1 \text{ gallon}/128 \text{ oz} \div$
43 $100 \text{ gallons}) \times 5\text{-}10 \text{ gallons/tree}$]. The product for the Kaiso 24WG granular formulation
44 (24% a.i.) indicates that 2.67 oz (avoirdupois) should be used per 100 gallons of water

1 and that 5-10 gallons should be applied per tree, which corresponds to an application rate
2 of almost exactly 0.002-0.004 lb a.i./tree [(2.67 oz x 1 lb/16 oz x 0.24 a.i. / 100 gallons) x
3 5-10 gallons/tree = 0.0020025-0.004005 lb a.i./tree].
4

5 The maximum annual application rate of 0.5 lb a.i./acre applies to tree applications as
6 well as airblast applications. Thus, at application rates of 0.002-0.004 lb a.i./tree, 125-
7 250 tree applications per acre would be allowed. If tree applications were repeated as in
8 broadcast applications—i.e., three applications per year at 2- to 4-week intervals—then
9 41-83 trees could be treated per acre over the course of 1 year.

10 ***2.3.3. Application Rates Used in Risk Assessment***

11 At the time this risk assessment was prepared, the Forest Service had not made a final
12 determination on the application rates, number of applications, and application intervals.
13 Initially, three applications at an application rate of 0.16 lb a.i./acre with a 3-week
14 interval were considered. The application rate of 0.16 lb a.i./acre was selected because
15 this is the recommended application rate for the control of coneworm and seed bug
16 species. The application interval of 3 weeks was selected based on information from the
17 Forest Service indicating that intervals of 2-4 weeks were under consideration (Bakke
18 2009a). Three applications were selected so that the cumulative application rate would
19 not exceed the labeled cumulative application rate of 0.5 lb a.i./acre.
20

21 Subsequent information from the Forest Service indicated that the application rate of 0.16
22 lb a.i./acre could exceed the application rate that the Forest Service might use and that
23 past practice with esfenvalerate had involved 6 applications at 2-week intervals (Bakke
24 2009c). This assessment is consistent with other uses of lambda-cyhalothrin in Forest
25 Service programs—i.e., an application rate of 0.025 lb a.i./acre in Region 8 (Mistretta
26 2007). Consequently, for the current Forest Service risk assessment, six applications at
27 0.08 lb a.i./acre at 2-week intervals are used for the exposure assessments at the Chico
28 and Foresthill sites. As discussed further in Section 3.2.3.4.3 (GLEAMS Modeling),
29 other application rates are considered (e.g., three applications at 0.16 lb a.i./acre with a 4-
30 week application interval); however, these variations do not have a substantial impact on
31 the estimated concentrations of lambda-cyhalothrin in water. Varying application rates
32 for lambda-cyhalothrin are also discussed in the risk characterization for human health
33 (Section 3.4) and ecological effects (Section 4.4).
34

35 Applications associated with other potential uses of lambda-cyhalothrin in Forest Service
36 programs may vary considerably. Lambda-cyhalothrin is incorporated into the Forest
37 Service's WorksheetMaker program, which allows for any number of applications at any
38 interval. For WorksheetMaker, the WRC values are based on six applications at 2 week
39 intervals. If a different application series is used in Forest Service programs, estimates of
40 water contamination rates could be developed based on the application series and
41 application site under consideration. As discussed further in Section 3.2.3.4.3.2 (Results
42 of Site-Specific Modeling), differences in the characteristics of the application site appear
43 to be more significant than differences in the application schedule or application rates
44 over the range of 0.08-0.16 lb a.i./acre.

1 **2.5. USE STATISTICS**

2 Forest Service pesticide use reports do not include information on lambda-cyhalothrin
3 (<http://www.fs.fed.us/foresthealth/pesticide/reports.shtml>), because it has not been used
4 in Forest Service programs. Nonetheless, the proposed uses of lambda-cyhalothrin
5 appear to be very minor compared with other uses of this insecticide in California and
6 many parts of the United States.

7
8 As noted in Section 1, the USDA Forest Service is considering the use of lambda-
9 cyhalothrin at only two sites in California—i.e., the Chico site (Chico Genetic Resources
10 and Conservation Center located in the Mendocino National Forest) and the Foresthill
11 site (Foresthill Genetics Center located in the Tahoe National Forest). The Foresthill site
12 consists of 118 acres of seed production trees, and the Chico site consists of 83.7 acres of
13 seed production trees. Thus, the Forest Service would treat a maximum of 201.7 acres in
14 a given season. At the maximum annual application rate of 0.5 lb a.i./acre, the maximum
15 amount of lambda-cyhalothrin that the Forest Service would use is about 100 lbs a.i.
16 [201.7 acres x 0.5 lb a.i./acre = 100.85 lbs a.i.].

17
18 By comparison, a total of 59,505 acres were treated with lambda-cyhalothrin in
19 California during 2007, the most recent year for which data are available from the
20 California Department of Pesticide Regulation (CDPR 2008, p. 84). A total of 31,633
21 pounds were applied in agricultural applications in California (CDPR 2008, p. 219). In
22 terms of the number of acres treated, lambda-cyhalothrin was the most widely used
23 insecticide in California in 2007. The increased use of lambda-cyhalothrin may be
24 associated with the upcoming patent expiration for this insecticide. The price of
25 treatment with lambda-cyhalothrin is only about \$3.00/acre (CDPR 2008, p. 84).

26
27 The USGS use statistics for lambda-cyhalothrin are illustrated in Figure 1. In 2002, the
28 most recent year for which USGS (2003) provides statistics, a total of about 235,000 lbs
29 of lambda-cyhalothrin was used in the continental United States in agricultural
30 applications. Lambda-cyhalothrin was most commonly applied to soybeans (≈19%),
31 cotton (≈18%), corn (≈13%), sweet corn (≈12%), and rice (≈10%). The greatest
32 concentrations of lambda-cyhalothrin agricultural applications occurred in the Southeast
33 (Forest Service Region 8), Northeast (Forest Service Region 9), western South Dakota (in
34 Forest Service Region 2), and central California (Forest Service Region 5).

35
36 Based on the above use summary, the uses of lambda-cyhalothrin under consideration by
37 the Forest Service at the two sites in California are very minor relative to agricultural
38 uses in California ($89.35 \text{ lbs} \div 31,633 \text{ lbs} \approx 0.0028 = 0.28\%$) or total agricultural uses in
39 the United States ($89.35 \text{ lbs} \div 235,000 \text{ lbs} \approx 0.00038 = 0.038\%$).

40 **2.6. SITE DESCRIPTIONS**

41 The characteristics of the Chico and Foresthill sites are summarized in Table 4. Most of
42 the data given in Table 4 are related to the Gleams-Driver modeling, as discussed further
43 in Section 3.2.3.4.3. An aerial view of the Chico site is given in Figure 2, and the
44 corresponding view for the Foresthill site is given in Figure 3. Outputs from AgDrift are

1 included beneath the site maps in Figures 2 and 3. These are discussed further in Section
2 3.2.3.4.3.

3
4 The site maps in Figures 2 and 3 are taken from the USDA Soil Survey website
5 (<http://websoilsurvey.nrcs.usda.gov/app>) based on latitude and longitude coordinates
6 provided by the Forest Service. Most of the data given in Table 4 are also taken from the
7 USDA Soil Survey website, except as otherwise noted in the footnotes to Table 4. Each
8 site is briefly summarized in the following paragraphs.

9 **2.6.1. Chico Site**

10 The Chico Genetic Resources and Conservation Center is located in the Mendocino
11 National Forest near the outskirts of Chico, California. As summarized in Table 4, the
12 site covers a 203-acre area within which 83.7 acres would be treated with pesticides. As
13 illustrated in Figure 2, the Chico site is in a populated area with residences located about
14 0.3 miles to the east and west of the site. Based on images from Google Earth, the site
15 appears to be open to and used by the general public. The soil at the Chico site consists
16 primarily of fine sandy loam with a 0 to 1 % slope—i.e., the area designated as 447 in
17 Figure 2.

18
19 Comanche Creek runs through the center of the site, and Butte Creek/Diversion Channel
20 runs along the eastern boarder of the site. A roadway open to the public appears to
21 parallel Comanche Creek. Flow to Comanche Creek is controlled by the Oakee Dam in
22 Butte Creek. During the winter months, water flow to Comanche Creek is turned off and
23 water flow in Comanche Creek occurs only during storms due to runoff water from the
24 Comanche Creek watershed. Nonetheless, standing water and near surface flows may be
25 found in Comanche Creek year round. The maximum flow in Comanche Creek is
26 estimated to be 150 cubic feet per second (USDA/FS 1998). The maximum flow rate of
27 150 cubic feet per second corresponds to approximately 367,000,000 liters/day [150
28 cubic feet per second x 28.32 L/cubic foot x 60 sec/min x 60 min/hr x 24 hr/day \approx 3.67 x
29 10⁸ L/day].

30 **2.6.2. Foresthill Site**

31 The Foresthill Genetics Center is located in the Tahoe National Forest. As summarized
32 in Table 4, the Foresthill site covers a 342-acre area within which 45 acres would be
33 treated with pesticides. Unlike the Chico site, the Foresthill Genetics Center is in a
34 relatively remote location. The nearest inhabited area appears to be the city of Foresthill
35 which is about 6 miles to the southwest of the Foresthill site.

36
37 The soils in the Foresthill site consist largely of well-drained loam. The topography of
38 the Foresthill site is much more variable than that of the Chico site, with slopes ranging
39 from 2 to 50% with a representative slope of about 16% (<http://websoilsurvey.nrcs.usda.gov/app>).

40
41
42 A portion of McBride Creek is within the Foresthill site. In terms of the current Forest
43 Service risk assessment, the most important characteristic of McBride Creek is that it
44 does not contain fish within the reach encompassed by the Foresthill site (Bakke 2010).

1 Based on elevations and imagery from both the USDA Soil Survey website and Google
2 Earth it appears that McBride Creek may originate in the Foresthill site and flow to the
3 southwest out of the Foresthill site. Unlike Comanche Creek, flows in the McBride
4 Creek appear to be unregulated. No information is available on flow rates; however,
5 according to the Forest Service, McBride Creek is ephemeral, with significant flows from
6 December to May (USDA/FS 1998). It appears likely that flows might occur at any time
7 as a result of atypical storm events; however, this speculation has not been confirmed by
8 the Forest Service. As discussed in Section 3.2.3.4.3, there is a 200-foot, heavily
9 vegetated buffer between the stream and areas likely to be treated with pesticides.

1 **3.1. HAZARD IDENTIFICATION**

2 **3.1.1. Overview**

3 Lambda-cyhalothrin is a pyrethroid insecticide that interferes with the normal functioning
4 of nerve cells. Most of the critical information on the hazard identification for lambda-
5 cyhalothrin comes from reviews of studies submitted to U.S. EPA in support of the
6 registration for lambda-cyhalothrin as well as a more general review by ATSDR on the
7 toxicity of pyrethroid insecticides to mammals.

8
9 U.S. EPA's Office of Pesticide Programs (U.S. EPA/OPP) classifies potential acute
10 hazards, based on several standard tests, ranging from the most hazardous (Category I) to
11 the least hazardous (Category IV). U.S. EPA/OPP reviewed the acute toxicity data on
12 lambda-cyhalothrin and classified it as Category II (moderately toxic), based on acute
13 oral, dermal and inhalation toxicity; Category II based on eye irritation (i.e., a moderate
14 eye irritant); and Category IV based on skin irritation (not a skin irritant). In addition, the
15 EPA does not consider lambda-cyhalothrin a skin sensitizer; nonetheless, dermal
16 exposure to lambda-cyhalothrin as well as many other pyrethroids may cause numbness
17 or tingling of the skin, a condition commonly referred to as paresthesia.

18
19 Lambda-cyhalothrin is neurotoxic; however, neurotoxicity is not always the most
20 sensitive endpoint in longer-term exposures. Weight loss or decreased body weight gain
21 are the effects commonly observed at doses below those associated with frank signs of
22 neurotoxicity. Changes in body weight gain can be mediated by the endocrine system,
23 and some studies suggest that lambda-cyhalothrin may affect normal endocrine function.
24 Several other pyrethroids also affect the endocrine system; however, it is not clear whether
25 these effects are direct or secondary to effects on the nervous system. Lambda-
26 cyhalothrin is not classified as carcinogenic by the EPA.

27 **3.1.2. Mechanism of Action**

28 Lambda-cyhalothrin is a pyrethroid, which is a class of man-made insecticides which are
29 structurally similar to pyrethrins, a group of naturally occurring insecticides. The
30 primary site of action for both pyrethrins and pyrethroids is the voltage-gated membrane
31 sodium channel of nerve cells. The basic function of nerve cells involves repeated
32 polarization and depolarization associated with neural activation or firing. These
33 processes are controlled by channels which allow for the influx of ions into nerve cells.
34 Both pyrethroids and pyrethrins inhibit the closing of sodium channels and thus disrupt
35 normal nerve function. Only about 0.6% of the sodium channel gates need to be affected
36 into order to elicit signs of neurotoxicity (ATSDR 2003). Some pyrethroids may also
37 alter chlorine channels; however, this does not appear to be the case with lambda-
38 cyhalothrin (Burr and Ray 2004).

39
40 Based on chemical structure, pyrethroids are classified as Type I pyrethroids (compounds
41 with no cyano group) or Type II pyrethroids (compounds with a cyano group). As
42 illustrated in Section 2.2, lambda-cyhalothrin contains a cyano group (i.e., a carbon-
43 nitrogen triple bond) and is classified as a Type II pyrethroid. Type I and Type II

1 pyrethroids differ in signs of neurotoxicity. Type I pyrethroids typically induce fine
2 tremors, increased body temperatures, and coma. Type II pyrethroids induce involuntary
3 movements, salivation, enhanced responses to stimuli, and coarse body tremors (ATSDR
4 2003). Signs of neurotoxicity after exposure to lambda-cyhalothrin are consistent with
5 the signs of toxicity typically associated with Type II pyrethroids (Hossain et al. 2005);
6 moreover, behavioral excitation has been observed in some humans following over-
7 exposure to lambda-cyhalothrin (Martinez-Larrantildeaga et al. 2003). In severely
8 poisoned animals, including humans, a broader range of effects may develop, the most
9 characteristic of which appear to be diarrhea, nausea, and vomiting (ATSDR 2003;
10 Hossain et al. 2005).

11
12 As indicated in Section 1, the Forest Service is considering the use of lambda-cyhalothrin
13 as an alternative to esfenvalerate, another Type II pyrethroid, for the control of coneworm
14 and seed bug species in two seed orchards. In addition to lambda-cyhalothrin and
15 esfenvalerate, commonly used Type II pyrethroids include gamma-cyhalothrin,
16 cyfluthrin, cypermethrin, deltamethrin, fenvalerate, fenpropathrin, flucythrinate,
17 flumethrin, fluvalinate, and tralomethrin (ATSDR 2003).

18
19 Lambda-cyhalothrin causes a spectrum of other biochemical effects that are generally
20 consistent with oxidative damage (El-Demerdash 2007; Fetoui et al. 2009). While
21 lambda-cyhalothrin does not uncouple oxidative phosphorylation, it does affect
22 mitochondrial respiration (Gassner et al. 1997). Generally, most biochemical effects of
23 exposure to pyrethroids not directly associated with neurotoxicity and are considered
24 secondary effects (ATSDR 2003).

25 ***3.1.3. Pharmacokinetics and Metabolism***

26 ***3.1.3.1. General Considerations***

27 Very little information is available on the pharmacokinetics of lambda-cyhalothrin. The
28 only human data come from the occupational exposure study by Leng et al. (1997) which
29 reports an average plasma half-life of 6.4 hours for lambda-cyhalothrin as well as several
30 other pyrethroids. Anadon et al. (2006) studied the pharmacokinetics of lambda-
31 cyhalothrin in rats following both intravenous and oral exposures. By either route of
32 exposure, lambda-cyhalothrin was widely distributed, with the highest concentrations
33 detected in the hypothalamus and the myenteric plexus—i.e., an area of unmyelinated
34 fibers enervating the gastrointestinal tract. The plasma half-lives in rats were 8.55 hours
35 after intravenous administration and 14.43 hours after oral administration. Whole body
36 elimination half-lives were 7.55 hours after intravenous exposure and 10.27 hours after
37 oral exposure, which indicates that the excretion of lambda-cyhalothrin is as rapid in rats
38 as in humans. Consistent with the mechanism of action of lambda-cyhalothrin and other
39 pyrethroids, half-lives in nerve tissues were substantially greater (12-34 hours) than half-
40 lives in plasma.

41 ***3.1.3.2. Absorption***

42 The available literature on lambda-cyhalothrin does not include information on dermal
43 absorption. U.S. EPA/HED (1997c, p. 8; 2002, p. 12) summarizes the results of a dermal

1 exposure study in humans in which the average first-order dermal absorption rate was
2 0.0012 day^{-1} with a range from 0.0004 to 0.0019 day^{-1} . These values correspond to
3 hourly dermal absorption rates of about 0.00005 (0.000017 - 0.000079) hour^{-1} . In the
4 most recent EPA hazard identification for lambda-cyhalothrin, U.S. EPA/HED (2002)
5 uses a conservative estimate of 0.01 day^{-1} or about $0.00042 \text{ hour}^{-1}$ for exposure
6 assessments.

7
8 In the absence of experimental data, Forest Service risk assessments generally adopt
9 estimates of dermal absorption rates based on quantitative structure activity relationships
10 (QSAR), as documented in SERA (2007a). Using these methods with the molecular
11 weight (449.9 g/mole) and K_{ow} ($10,000,000$) for lambda-cyhalothrin, the estimated first-
12 order dermal absorption rates are approximately 0.0039 (0.00073 – 0.021) hour^{-1} . The
13 calculation of these rates is detailed in Worksheet B06 in the EXCEL workbooks that
14 accompany this risk assessment. Notably, the lower bound of the QSAR estimates—i.e.,
15 $0.00073 \text{ hour}^{-1}$ —is about 10 times greater than the upper bound of the dermal absorption
16 rates from the human study—i.e., $0.000079 \text{ hour}^{-1}$. Confidence in the QSAR estimates is
17 limited also because the algorithm is based on an analysis of compounds with K_{ow} values
18 ranging up to only about $3,000,000$ and molecular weights up to 400 g/mole . Although
19 the QSAR estimates of the first-order dermal absorption rates are substantially higher
20 than those from the human study, the measured rates from the human study are clearly
21 preferable to the QSAR estimates for use in the risk assessment.

22
23 For the current Forest Service risk assessment, the first-order dermal absorption rates are
24 taken as 0.00005 (0.000017 - 0.00042) hour^{-1} or about 0.0012 (0.00041 - 0.01) day^{-1} . Note
25 that the central estimate and the lower bound are taken directly from the human study
26 summarized by U.S. EPA. The upper bound, however, is adopted from the dermal
27 absorption rate used by U.S. EPA/HED (2002), 0.01 day^{-1} or about $0.00042 \text{ hour}^{-1}$.
28 These dermal absorption rates are applied to exposure scenarios involving dermal contact
29 with contaminated vegetation as well as spills of the pesticide onto the skin. In these
30 types of exposure scenarios, the assumption of first-order dermal absorption is
31 appropriate (SERA 2007a).

32
33 Another set of exposure scenarios used in this risk assessment involves the assumption of
34 zero-order absorption (i.e., the dermal absorption rate is constant over time). This type of
35 assumption is reasonable when the skin is in constant contact with an amount or
36 concentration of the pesticide and is fundamental to exposure scenarios in which workers
37 wear contaminated gloves. This scenario assumes that the amount of pesticide saturating
38 the inside of the gloves is greater than the degree of dermal absorption. When
39 experimental data are not available to estimate a zero-order dermal absorption rate (i.e.,
40 typically referred to as a K_p in units of cm/hour), Forest Service risk assessments
41 generally use a QSAR algorithm developed by the EPA (U.S. EPA/ORD 1992). This
42 approach is discussed in further detail in SERA (2007a). As detailed in Worksheet B05
43 of the EXCEL workbooks that accompany this risk assessment, the QSAR algorithm
44 developed by the EPA results in an estimated zero-order dermal absorption rate of 0.28
45 (0.073 - 1.1) cm/hour . As with the QSAR estimates of the first-order dermal absorption
46 rates, confidence in the estimated zero-order rates is limited because the algorithm used

1 to estimate these rates is based on data from compounds with K_{ow} values of up to only
2 about 320,000, which is about 31 times less than the lambda-cyhalothrin K_{ow} of
3 10,000,000.

4
5 Given the discrepancies between the experimental and QSAR estimates of the first-order
6 dermal absorption rates for lambda-cyhalothrin, confidence in the zero-order rates
7 estimated from the EPA algorithm is extremely low. As discussed above, the lower
8 bound of the QSAR estimates for the first-order rates appear to overestimate the rates by
9 at least a factor of 10. For the current Forest Service risk assessment, the zero-order
10 dermal absorption rate of 0.28 (0.073-1.1) cm/hour is adjusted downward by a factor of
11 10 and rounded to one significant place—i.e., rates of 0.03 (0.007-0.1) cm/hour. The
12 uncertainties in the zero-order dermal absorption rates are substantial, as discussed
13 further in the risk characterization (Section 3.4).

14 **3.1.3.3. Excretion**

15 Although excretion rates are not used directly in either the dose-response assessment or
16 risk characterization, excretion half-lives can be used to infer the effect of longer-term
17 exposures on body burden, based on the *plateau principle* (e.g., Goldstein et al. 1974).
18 The concentration of the chemical in the body after a series of doses (X_{Inf}) over an
19 infinite period of time can be estimated based on the body burden immediately after a
20 single dose, X_0 , by the relationship:

$$21 \quad \frac{X_{Inf}}{X_0} = \frac{1}{1 - e^{-kt^*}}$$

22
23
24 where t^* is the interval between dosing and k is the first-order excretion rate. As
25 discussed in Section 3.1.3.1, the whole body half-life of lambda-cyhalothrin in rats after
26 oral administration is 10.27 hours or about 0.42 days. Based on the assumption of first-
27 order excretion, k may be estimated from the half-life ($T_{1/2}$) as:

$$28 \quad k = \ln(2) \div T_{1/2}$$

29
30
31
32 Based on this relationship, the half-life of 0.42 days corresponds to an elimination rate (k)
33 of 1.65 day^{-1} [$\ln(2) \div 0.42 \text{ days}$]. Substituting this value into the above equation for the
34 plateau principle, the long-term body burden relative to the single dose body burden
35 would be about 1.24. In other words, the available pharmacokinetic data on lambda-
36 cyhalothrin do not suggest that prolonged exposure will result in substantial accumulation
37 of the insecticide. This supposition is further supported by the ATSDR general review of
38 the pharmacokinetics of pyrethroids (ATSDR 2003), which indicates that the excretion of
39 most pyrethroids is relatively rapidly in mammals.

40 **3.1.4. Acute Oral Toxicity**

41 U.S. EPA's Office of Pesticide Programs (U.S. EPA/OPP) classifies potential acute
42 hazards, based on several standard tests, ranging from the most hazardous (Category I) to
43 the least hazardous (Category IV). As summarized in various EPA reviews (Hurley

1 1989a; U.S. EPA/HED 1997c; 2002), the oral LD₅₀ for technical grade lambda-
2 cyhalothrin is 79 mg/kg bw in male rats and 56 mg/kg bw in female rats. These values
3 are used to classify lambda-cyhalothrin as Category II for acute oral toxicity.

4
5 As discussed by ATSDR (2003, p. 34), the acute oral toxicity of pyrethrins and
6 pyrethroids can be influenced by the dosing vehicle—i.e., the material in which the test
7 substance is dissolved prior to dosing the animals. In the example cited by ATSDR, the
8 LD₅₀ value for permethrin in corn oil (i.e., 584 mg/kg bw) was substantially lower than
9 the LD₅₀ value for permethrin administered without a vehicle (i.e., 3801 mg/kg bw). The
10 oral LD₅₀ values for various formulations of lambda-cyhalothrin suggest a similar pattern.
11 As summarized in Appendix 1, Table 1, the oral LD₅₀ values for the liquid formulations,
12 expressed in units of active ingredient, range from 7.2 to about 41 mg a.i./kg bw. The
13 LD₅₀ value for the one granular formulation is 74.4 mg a.i./kg bw, which falls within the
14 range of oral LD₅₀ values for technical grade lambda-cyhalothrin.

15 ***3.1.5. Subchronic or Chronic Systemic Toxic Effects***

16 The open literature does not include information on the subchronic or chronic toxicity of
17 lambda-cyhalothrin. U.S. EPA/OPP summarizes information on two subchronic studies
18 in rats and one subchronic study in mice (U.S. EPA/HED 1997c, p. 1 ff) as well as
19 chronic toxicity studies in rats, mice, and dogs (U.S. EPA/HED 1997c, p. 3 ff). The
20 summaries of these studies from U.S. EPA/HED (1997c) are the basis for the following
21 discussion.

22
23 The two subchronic studies in rats involved 90-day dietary exposures to Wistar derived
24 rats in which the most sensitive endpoint was body weight loss which occurred at a
25 dietary concentration of 250 ppm (≈12.4 mg/kg bw/day) with a NOEL of 50 ppm (≈2.5
26 mg/kg bw/day) in both studies. In one of these studies, a slight but statistically
27 significant decrease in food conversion efficiency was noted in female rats. The
28 subchronic study in mice was conducted over a 28-day period. The NOEC in mice was a
29 dietary concentration 500 ppm, corresponding to a dose of 64.2 mg/kg bw/day in males
30 and 77.9 mg/kg bw/day in females. At the next higher dietary concentration of 2000 ppm
31 (≈309 mg/kg bw/day in males and ≈294 mg/kg bw/day in females), there were signs of
32 neurotoxicity —i.e., abnormal gait and posture—and other effects of toxicity, including
33 weight loss, slight changes in hematology and organ weights. Because the duration of the
34 study in mice was substantially less than the duration of the studies in rats, it is not clear
35 whether the substantially higher NOEL in mice, relative to rats, reflects species
36 difference, the effect of duration, or a combination of these and other factors.

37
38 The chronic studies in rats and mice both involved a 2-year period of exposure and
39 suggest that mice may be somewhat more tolerant than rats to dietary administration of
40 lambda-cyhalothrin. The rat study yielded a dietary NOEL of 50 ppm (corresponding to
41 a dose of 2.5 mg/kg bw/day) with a corresponding LOAEL of 250 ppm (12.5 mg/kg
42 bw/day) based on decreased body weight with no signs of neurotoxicity. The NOEL and
43 LOAEL values are identical to those in the subchronic rat studies. In the chronic study in
44 mice, the dietary NOEL was 100 ppm (15 mg/kg bw/day) with a LOAEL of 500 ppm (75

1 mg/kg bw/day). The LOAEL for mice is also based on decreased body weight, although
2 piloerection and abnormal posture were observed in some test animals.

3
4 The chronic study in dogs is based on the administration of lambda-cyhalothrin in gelatin
5 capsules at doses of 0.1, 0.5, or 3.5 mg/kg bw/day for 1 year. No adverse effects were
6 noted at the lowest dose. At 0.5 mg/kg bw/day, signs of neurotoxicity (abnormal gait)
7 were noted in some animals over the period from Week 2 through Week 9 of the study.
8 As discussed further in Section 3.3 (Dose-Response Assessment), the dose of 0.5 mg/kg
9 bw is classified as a LOAEL for chronic exposure but a NOAEL for acute exposure. At
10 3.5 mg/kg bw/day, signs of neurotoxicity (ataxia, tremors, convulsions, and vomiting)
11 were noted during the first 2 weeks of the study. As also discussed in Section 3.3.2, the
12 dose of 3.5 mg/kg bw/day is classified as the LOAEL for acute exposure in the derivation
13 of the acute RfD.

14 **3.1.6. Effects on Nervous System**

15 As discussed in ATSDR (2003), lambda-cyhalothrin and many other pyrethroids as well
16 as pyrethrins are clearly neurotoxic, and the mechanism of neurotoxicity is understood
17 relatively well (Section 3.1.2). Wolansky et al. (2006) assayed the acute neurotoxicity of
18 both lambda-cyhalothrin and esfenvalerate to rats. In this study, estimates were made of
19 both the NOEC or threshold dose for neurotoxicity as well as the dose associated with a
20 30% decrease in motor activity in a maze (EC₃₀) after gavage dosing. Lambda-
21 cyhalothrin and esfenvalerate showed similar potencies with NOEC values of 0.52 and
22 0.48 mg/kg bw, respectively, and EC₃₀ values of 1.32 and 1.2 mg/kg bw, respectively.
23 The acute NOEC of 0.52 mg/kg bw in rats reported by Wolansky et al. (2006) is virtually
24 identical to the estimated acute NOEC for neurotoxicity in dogs (Section 3.1.5). The
25 NOEC of 0.5 mg/kg bw is also supported by the NOEC in rats of 1 mg/kg bw/day over a
26 7-day period of exposure (Righi and Palermo-Neto 2003). Consistent with the rapid
27 excretion of lambda-cyhalothrin (Section 3.1.3), signs of neurotoxicity associated with
28 sublethal doses of lambda-cyhalothrin as well as several other pyrethroids are reversible,
29 with recovery times as short as 3-4 hours after dosing (Wright et al. 1988).

30
31 While lambda-cyhalothrin is clearly neurotoxic, neurotoxicity is not always the most
32 sensitive endpoint in standard toxicity studies. As summarized in Section 3.1.5,
33 subchronic and chronic studies in rats and mice indicate that decreased body weight is a
34 more sensitive endpoint than neurotoxicity in these species. The same pattern was
35 observed in a delayed neurotoxicity study in which the most sensitive endpoint in hens
36 was decreased body weight absent any signs of neurotoxicity (U.S. EPA/HED 2002). In
37 a specialized test for acute neurotoxicity in rats, however, clear signs of neurotoxicity
38 (i.e., piloerection, ataxia, salivation, lacrimation, and decreased motor activity) were
39 noted at doses of 35 mg/kg bw/day and higher, with a NOAEL of 10 mg/kg bw (U.S.
40 EPA/HED 2002, p.4). As discussed further in Section 3.1.9 (Reproductive and
41 Developmental Effects), signs of neurotoxicity in rats were also noted in a developmental
42 study.

1 **3.1.7. Effects on Immune System**

2 The most recent EPA review of lambda-cyhalothrin (U.S. EPA/HED 2002) does not
3 address its potential to affect the immune system. Concern for the effects of pyrethroids
4 on immune function is raised by ATSDR:

5
6 *Results of a few recent animal studies suggest that*
7 *neurodevelopmental, reproductive, and immunological effects may*
8 *result following exposure to some pyrethroids at levels below those*
9 *that induce overt signs of neurotoxicity.*

10 ATSDR 2002, p. 16

11
12 Krishnappa et al. (1999) exposed two groups of rats to a very high dietary concentration
13 (3000 ppm) of an unspecified 2.5% EC (emulsifiable concentrate) formulation of lambda-
14 cyhalothrin for 90 days to assay its effects on immune function. One test group was
15 treated with *Brucella abortus* antigen, while the other test group was not. No effects on
16 immune function were noted in the antigen-free rats; however, in the antigen-treated
17 group there were significant decreases in total immunoglobulin concentration, total
18 leukocyte count, and total lymphocyte count. Based on the subchronic toxicity studies
19 summarized in Section 3.1.5, a dietary concentration of 3000 ppm would be expected to
20 cause signs of toxicity in rats. Nevertheless, Krishnappa et al. (1999) do not report signs
21 of toxicity other than changes in immune response.

22
23 Righi and coworkers (Righi and Palermo-Neto 2005; Righi et al. 2009) assayed the
24 immunotoxicity of cyhalothrin—i.e. a mixture of four cyhalothrin isomers rather than the
25 two cyhalothrin isomers in lambda-cyhalothrin. Both studies report a decrease in
26 macrophage activity which was statistically significant after *in vivo* 7-day exposures to
27 doses of 1 and 3 mg/kg bw/day but not at 0.6 mg/kg bw/day. While these studies are not
28 directly applicable to lambda-cyhalothrin, the NOAEL of 0.6 mg/kg bw/day is somewhat
29 higher than the acute NOAEL of 0.5 mg/kg bw/day for neurotoxicity in dogs exposed to
30 lambda-cyhalothrin (Section 3.1.5).

31 **3.1.8. Effects on Endocrine System**

32 Impacts of pesticides on endocrine function are often assessed indirectly based on
33 standard toxicity studies as well as reproduction studies (Section 3.1.9). Based on this
34 type of information, the EPA concludes that:

35
36 *There is no evidence that lambda-cyhalothrin induces any*
37 *endocrine disruption.*

38 U.S. EPA/HED 2002, p. 22

39
40 The ATSDR review (ATSDR 2003, pp. 107-108) indicates that several pyrethroids affect
41 endocrine function. Furthermore, there is some indication in the open literature that
42 lambda-cyhalothrin may affect endocrine function.

43
44 In a 21-day gavage study in which rats were administered lambda-cyhalothrin at a dose of
45 approximately 0.73 mg/kg bw/day, serum triiodothyronine (T₃) and thyroxine (T₄) as

1 well as T₃/T₄ ratios were significantly suppressed and serum thyroid stimulating hormone
2 (TSH) levels were significantly increased. The dosing did not affect body weight gain,
3 and no other signs of toxicity were noted (Akhtar et al. 1996). In an *in vivo* study,
4 Ratnasooriya et al. (2003) exposed pregnant rats by gavage to Icon, a formulation of
5 lambda-cyhalothrin used in Sri Lanka, at doses of 6.3, 8.3, or 12.5 mg a.i./kg bw/day for
6 7 days. The primary adverse reproductive effect was increased pre-implantation losses.
7 This effect was blocked by co-administration of progesterone. No effects of lambda-
8 cyhalothrin were noted on birth weight, fetal morphology, pre-natal development, and
9 other standard reproductive parameters. Zhao et al. (2008) assayed the estrogenic effect
10 of lambda-cyhalothrin in a breast carcinoma cell line. Concentrations as low as 10⁻⁷ M
11 (about 45 µg/L) promoted cell proliferation—i.e., mimicked the effect of estrogen—and
12 the cell proliferation was blocked by the addition of an estrogen receptor antagonist at a
13 concentration of 10⁻⁹ M. Both of these observations suggest that lambda-cyhalothrin may
14 have estrogenic activity.

15 **3.1.9. Reproductive and Developmental Effects**

16 **3.1.9.1. Developmental Studies**

17 Developmental studies are used to assess whether a compound has the potential to cause
18 birth defects—also referred to as teratogenic effects—as well as other effects during
19 development or immediately after birth. These studies typically entail gavage
20 administration to pregnant rats or rabbits on specific days of gestation. Teratology assays
21 as well as studies on reproductive function (Section 3.1.9.2) are generally required for the
22 registration of pesticides. Very specific protocols for developmental studies are
23 established by U.S. EPA/OPPTS and are available at
24 http://www.epa.gov/opptsfrs/publications/OPPTS_Harmonized.

25
26 The EPA requires developmental studies for pesticide registration; accordingly, several
27 developmental studies are summarized in the most recent EPA/OPP review of lambda-
28 cyhalothrin (U.S. EPA/HED 2002, p. 5 ff). In both rats and rabbits, no signs of toxicity
29 were noted at doses of 10 mg/kg bw/day. In rats, signs of neurotoxicity were evident in
30 dams at 15 mg/kg bw/day; there were no evident effects on the offspring—i.e., the
31 developmental NOAEL was 15 mg/kg bw/day. In rabbits, decreases in body weight and
32 food consumption were noted at 30 mg/kg bw/day. As with rats, however, no effects
33 were noted on offspring, and 30 mg/kg bw/day was classified as a developmental
34 NOAEL.

35
36 As discussed in Section 3.1.8 (Effects on Endocrine System), the developmental study by
37 Ratnasooriya et al. (2003) reports a significant increase in implantation losses at 8.3 and
38 12.5 mg/kg bw/day, with a NOAEL of 6.3 mg/kg bw/day. Unlike the rat developmental
39 study summarized by U.S. EPA, dams in the study by Ratnasooriya et al. (2003)
40 evidenced signs of neurotoxicity at all dose levels. As also noted in Section 3.1.8, the
41 Ratnasooriya et al. (2003) study involved the use of a formulation of lambda-
42 cyhalothrin—i.e., ICON, which is not being considered for use by the Forest Service. In
43 an earlier study on the same formulation, Ratnasooriya et al. (2002) report that relatively
44 low oral doses (i.e., about 6.3 and 10 mg/kg bw) of lambda-cyhalothrin were associated
45 with a decrease in mating behavior in male rats.

1
2 In two studies conducted in Brazil, dermal doses of about 10 mg/kg bw cyhalothrin are
3 associated with delayed development in rats (Da Silva Gomes et al. 1999a,b). It is not
4 clear, however, that the isomeric composition of the cyhalothrin corresponds to lambda-
5 cyhalothrin. In another study conducted in Brazil, Moniz et al. (1990) report that no
6 adverse developmental effects were observed in rats exposed to 200 ppm concentrations
7 of cyhalothrin in drinking water over a 21-day period of lactation. The rat pups were
8 followed to adulthood, and the authors observed latent behavioral effects, reported as
9 abnormal responses in avoidance tests, in 90-day-old rats. Again, however, it is not clear
10 that the cyhalothrin used in this study corresponds to lambda-cyhalothrin.

11
12 In a recent study conducted in Algeria, Lebaili et al. (2008) report evidence of testicular
13 damage in rats exposed to very high concentrations of lambda-cyhalothrin in drinking
14 water—i.e., about 15,000 or 23,000 ppm. In this study, the lambda-cyhalothrin was
15 formulated as Karate 2.5 EC. As noted in Section 2, the use of Karate is not under
16 consideration by the Forest Service in California. In addition, the exposures in this study
17 appear to represent doses that are far higher than the NOEC values on which the RfD for
18 lambda-cyhalothrin is based. Lebaili et al. (2008) do not provide water consumption data
19 for the rats; they indicate that the body weights of the rats ranged from 225 to 250 g. The
20 allometric relationship for water consumption in mammals developed by U.S. EPA/ORD
21 (1993, Eq. 3-17, p. 3-10) is:

$$W_{(L)} = 0.099 B^{0.9}_{(kg)}$$

22
23
24
25 where **W** is the water consumption in liters and **B** is the body weight in kilograms. Thus,
26 a 0.225 kg rat would consume about 0.02475 liters of water. If the water contained
27 15,000 ppm (mg/L) of lambda-cyhalothrin, the rat would consume about 371 mg for a
28 total dose of about 1650 mg/kg bw. As discussed further in Section 3.3.2, the acute RfD
29 is based on an NOEC of 0.5 mg/kg bw/day, a factor of over 8000 higher than the lowest
30 dose used by Lebaili et al. (2008).

31
32 The discrepancies between the EPA review of the registrant-submitted developmental
33 studies (U.S. EPA/HED 2002) and the reports of developmental toxicity in the open
34 literature cannot be fully resolved. According to ATSDR (2002, p. 62 ff), developmental
35 toxicity is not characteristic of pyrethroids. At least some of the difference between the
36 studies in the open literature and the EPA studies may reflect differences in the isomeric
37 composition of the material tested or the inerts used in some of the formulations tested.

38 **3.1.9.2. Reproduction Studies**

39 Reproduction studies involve exposing one or more generations of the test animal to a
40 chemical compound. Generally, the experimental method involves dosing the parental
41 (P₁ or F₀) generation (i.e., the male and female animals used at the start of the study) to
42 the test substance prior to mating, during mating, after mating, and through weaning of
43 the offspring (F₁). In a 2-generation reproduction study, this procedure is repeated with
44 male and female offspring from the F₁ generation to produce another set of offspring (F₂).
45 During these types of studies, standard observations for gross signs of toxicity are made.

1 Additional observations often include the length of the estrous cycle, assays on sperm and
2 other reproductive tissue, and the number, viability, and growth of offspring.

3
4 As with developmental studies, the EPA requires at least one reproduction study for
5 pesticide registration. U.S. EPA/HED (2002) summarizes the results of one 3-generation
6 reproduction study in rats at doses of 0, 0.5, 1.5, or 5 mg/kg bw/day. This study is not
7 explicitly identified by author/date citation or by MRID number. The only adverse effect
8 observed was a decrease in adult and fetal body weight at 5 mg/kg bw/day. Thus, 1.5
9 mg/kg bw/day was classified as the NOAEL for both parents and offspring. Because no
10 effects were observed in reproductive parameters, 5 mg/kg bw/day was classified as the
11 developmental NOAEL.

12
13 The U.S. EPA's Office of Research and Development (U.S. EPA/ORD 1988) maintains a
14 database of reference doses (RfDs). This database also summarizes the results of a
15 reproduction study in rats using Karate. This study is identified as MRID 00154802 and
16 is referenced to Coopers Animal Health, Inc and Imperial Chemical Industries with a
17 study date of 1984. As discussed in Section 2.2, Karate is a commercial formulation of
18 lambda-cyhalothrin. U.S. EPA/ORD (1988) indicates that the dietary concentrations of
19 lambda-cyhalothrin used in this study were 0, 10, 30, and 100 ppm. The mg/kg bw dose
20 conversions used by U.S. EPA/ORD (1988) correspond to the doses cited by U.S.
21 EPA/HED (2002)—i.e., 0, 0.5, 1.5, or 5 mg/kg bw/day. The summary by U.S.
22 EPA/ORD (1988) is consistent with that of U.S. EPA/HED (2002) in indicating that
23 adverse effects were observed at the highest dose. The very brief summary of the study
24 in U.S. EPA/ORD (1988), however, identifies 1.5 mg/kg bw/day as a LOAEL based on
25 reduced body weight gain in offspring during weaning. This is not consistent with the
26 summary by U.S. EPA/HED (2002, p.6) indicating that 1.5 mg/kg bw/day was a
27 NOAEL.

28
29 The available literature on lambda-cyhalothrin includes a cleared review of a
30 reproduction study in rats (Milburn et al. 1984). The reference for the review does not
31 specify an MRID number; however, it is clear that the study is associated with Imperial
32 Chemical Industries. The review specifies dietary concentrations of 0, 10, 30, and 100
33 ppm, and identifies a parental NOAEL of 10 ppm (0.5 mg/kg bw/day).

34
35 Full copies of the studies submitted to EPA are not available for review and could not be
36 used to conduct the current Forest Service risk assessment. Based on the above
37 summaries, it is not entirely clear whether the summaries refer to three different studies
38 or reflect differing interpretations of the same study.

39
40 A 3-generation reproduction study using technical grade cyhalothrin is published in the
41 Indian literature (Deshmukh 1992). As with some of the developmental studies
42 conducted in foreign countries, the isomeric composition of the cyhalothrin is not clearly
43 specified in the publication. The results of the study are generally similar to those of the
44 study summarized by the EPA. Mice were dosed at 2.5 or 5.0 mg/kg bw. While not
45 clearly specified in the publication, the dosing appears to have involved gavage

1 administration. The only adverse effect was a decrease in body weight only in the P₁
2 mice at both dose levels. No effects on reproductive parameters were observed.

3 **3.1.10. Carcinogenicity and Mutagenicity**

4 As reviewed by U.S. EPA/HED (2002, p. 16 ff), a relatively standard set of
5 carcinogenicity and mutagenicity studies were submitted to the EPA in support of the
6 registration of lambda-cyhalothrin. In a chronic feeding study in rats at doses of up to
7 12.5 mg/kg bw/day, no evidence of carcinogenic activity was noted. In a chronic feeding
8 study in mice at doses of up to 75 mg/kg bw/day, an increase in mammary tumors in
9 female mice was noted. The significance of this effect was classified as equivocal
10 because the incidence of mammary tumors in the matched control group was low,
11 compared with historical control groups. Eight mutagenicity studies are reviewed by
12 U.S. EPA/HED (2002, p. 18 ff). Five of the studies indicate no mutagenic activity and
13 the other three studies are classified as inconclusive because of issues associated with the
14 experimental designs of the studies. Based on this information, U.S. EPA/HED (2002)
15 classifies cyhalothrin as a Group D chemical, indicating that its potential carcinogenicity
16 is indeterminate.

17
18 The open literature on lambda-cyhalothrin does not include carcinogenicity bioassays.
19 Naravaneni and Jamil (2005) report that lambda-cyhalothrin was positive in a comet
20 assay (for strand breaks in DNA) using human lymphocyte cultures. Similarly, Celik et
21 al (2003, 2005a,b) report chromosome aberrations in rat bone marrow after
22 intraperitoneal injections as well as oral administration. A micronucleus assay in fish
23 also suggests that lambda-cyhalothrin concentrations as low as 0.001 µg/L may be
24 genotoxic (Campana et al. 1999).

25
26 In terms of a quantitative significance to the human health risk assessment,
27 carcinogenicity is an issue only if the data are adequate to support the derivation of a
28 cancer potency factor. Because of the equivocal nature of the carcinogenicity data on
29 lambda-cyhalothrin, U.S. EPA/OPP has not proposed or derived a cancer potency factor
30 of lambda-cyhalothrin. Thus, consistent with the EPA risk assessments, the current
31 Forest Service risk assessment does not quantitatively consider cancer risk as an endpoint
32 of concern.

33 **3.1.11. Irritation and Sensitization (Effects on the Skin and Eyes)**

34 As reviewed by U.S. EPA/OPP (U.S. EPA/HED 2002), lambda-cyhalothrin is classified
35 as an eye irritant (Category II) but not a skin irritant (Category IV). Moreover, lambda-
36 cyhalothrin (i.e., the two isomer mixture) is not classified as a skin sensitizer, which
37 means its toxicity differs from that of cyhalothrin (i.e., the four-isomer mixture) which is
38 a skin sensitizer.

39
40 Many pyrethroids induce paresthesia, a numbness or tingling of the skin which may also
41 be characterized as a feeling of burning or itching skin (ATSDR 2003; Martinez-
42 Larrantildeaga et al. 2003). Skin irritation consistent with paresthesia is documented in
43 workers handling lambda-cyhalothrin (Spencer and O'Malley 2006; Moretto 1991).

1 **3.1.12. Systemic Toxic Effects from Dermal Exposure**

2 Technical grade lambda-cyhalothrin is classified as a Category II compound for acute
3 dermal toxicity in rats with LD₅₀ values of 623 mg/kg bw for male rats and 696 mg/kg
4 bw for female rats (U.S. EPA/HED 2002). As discussed further in Section 3.1.14.1, these
5 dermal LD₅₀ values for technical grade lambda-cyhalothrin are consistent with the dermal
6 LD₅₀ values reported for the formulations of lambda-cyhalothrin explicitly considered in
7 the current Forest Service risk assessment (Appendix 1, Table 2).

8
9 U.S. EPA also summarizes a subchronic dermal toxicity study in which lambda-
10 cyhalothrin was applied in rats at dermal doses of 1 or 10 mg/kg/day, 6 hours/day, for 21
11 days. Another group of rats was treated at 100 mg/kg bw/day for 2 days but the dose was
12 then reduced to 50 mg/kg/day for the remainder of the 21-day study. Although no
13 adverse effects were noted at the 10 mg/kg bw/day dose, the high dose group showed
14 signs of neurotoxicity as well as decreased body weight.

15 **3.1.13. Inhalation Exposure**

16 As reviewed by U.S. EPA/HED (2002), technical grade lambda-cyhalothrin is classified
17 as a Category II compound for acute inhalation toxicity based on rat inhalation LC₅₀
18 values of 0.065 mg/L. U.S. EPA/HED (2002) also summarizes the results of a
19 subchronic inhalation study in rats in which the animals were exposed for 6 hours/day, 5
20 days/week for 3 weeks. No effects were noted at concentrations of 0.0003 mg/L. At
21 0.0033 mg/L adverse effects included signs of neurotoxicity and decreased body weight
22 gain (U.S. EPA/HED 2002, p. 14).

23
24 Moretto (1991) reports signs of respiratory irritation—i.e., irritation to the nose and throat
25 as well as coughing and sneezing—in workers involved in the indoor application of
26 lambda-cyhalothrin. Spencer and O'Malley (2006) report signs of respiratory irritation in
27 workers exposed to lambda-cyhalothrin as well as propargite and sulfur. Because of the
28 mixed exposure, it is not clear that lambda-cyhalothrin was the agent causing respiratory
29 irritation in these workers.

30 **3.1.14. Other Ingredients in Formulations and Adjuvants**

31 **3.1.14.1. Other Ingredients in Formulations**

32 The EPA is responsible for regulating other ingredients and adjuvants in pesticide
33 formulations. As implemented, these regulations affect only pesticide labeling and
34 testing requirements. The term *inert* was formerly used to designate compounds that do
35 not have a direct toxic effect on the target species. Although the term *inert* is codified in
36 FIFRA, some inerts may be toxic; therefore, the EPA now uses the term *Other*
37 *Ingredients* instead of the term *inerts*. This approach is adopted in the current risk
38 assessment.

39
40 As discussed in Section 2.2 and detailed in Table 3, only limited information is available
41 on the inerts in the lambda-cyhalothrin formulations explicitly considered in the current
42 Forest Service risk assessment. Several of the liquid formulations of lambda-cyhalothrin
43 contain naphthalene as well as other petroleum or aromatic solvents. All of these

1 compounds may be generally classified as petroleum distillates. Petroleum distillates are
2 highly diverse mixtures of aromatic and aliphatic hydrocarbons, and the specific blend of
3 aromatic and aliphatic hydrocarbons varies according to the distillation and refining
4 methods used. As reviewed by ATSDR (1999), petroleum distillates can induce a wide
5 range of toxic effects, particularly effects on the nervous system. U.S. EPA/OPP has not
6 yet completed their RED for aromatic petroleum hydrocarbons
7 (<http://www.epa.gov/pesticides/reregistration/status.htm>). The RED for aliphatic
8 petroleum hydrocarbons (U.S. EPA/OPP 2007c, p. 5 ff) notes little basis for concern for
9 the use of the aliphatic components of petroleum products as other ingredients in
10 pesticide formulations.

11
12 Given the complexity and variability of petroleum distillates as well as the limited
13 information available on the identity of the petroleum components in formulations of
14 lambda-cyhalothrin, it is difficult to assess the extent to which the other ingredients in
15 lambda-cyhalothrin formulations contribute to the toxicity of these formulations. One
16 approach to assessing this issue is to compare the toxicity of the formulations, expressed
17 in units of active ingredient, to the toxicity of the active ingredient itself. For lambda-
18 cyhalothrin, these comparisons yield differing results depending on the route of exposure.

19
20 As summarized in Appendix 1, Table 2, the oral LD₅₀ values for the lambda-cyhalothrin
21 formulations explicitly considered in this risk assessment are all less than the oral LD₅₀
22 value for technical grade lambda-cyhalothrin. The LD₅₀ values for the liquid
23 formulations range from about 7.2 mg a.i./kg bw (LambdaStar 1 CS) to 41 mg a.i./kg bw
24 (Warrior II). The single granular formulation, Kaiso 24 WG, has an oral LD₅₀ of 74.4
25 mg a.i./kg bw, within the range of oral LD₅₀ values reported for technical grade lambda-
26 cyhalothrin—i.e., 56 mg/kg for females and 79 mg/kg for males. Thus, the oral LD₅₀
27 values for the liquid formulations of lambda-cyhalothrin suggest that other ingredients
28 may contribute to the toxicity of these formulations.

29
30 Inhalation LC₅₀ values, however, display the opposite pattern. As summarized in
31 Appendix 1, Table 3, the 4-hour inhalation LC₅₀ values for lambda-cyhalothrin
32 formulations range from 0.071 mg/L (Lambda-Cy EC) to 0.711 mg/L (Warrior II);
33 whereas, the inhalation LC₅₀ for technical grade lambda-cyhalothrin is reported as 0.065
34 mg/L. This pattern suggests that the other ingredients in these lambda-cyhalothrin
35 formulations do not contribute to the toxicity of the formulations.

36
37 All dermal LC₅₀ values for lambda-cyhalothrin formulations are reported as ranging from
38 >288 mg/L to >1,200 mg/L. The *greater than* designations indicate that the LC₅₀ values
39 for the formulations could not be determined because of low rates of mortality. All of
40 these reported LC₅₀ values are consistent with the dermal LC₅₀ values of 632 mg/kg in
41 female rats and 696 mg/kg in male rats.

42 **3.1.14.2. Adjuvants**

43 Adjuvants may be used in applications of lambda-cyhalothrin formulations. The most
44 commonly recommended adjuvants are nonionic surfactants, once refined vegetable oil
45 concentrate, or methylated sunflower oils. The product labels for the formulations

1 specifically note that adjuvants should be used only if they contain ingredients
2 specifically approved by the EPA.

3
4 As with most Forest Service risk assessments as well as pesticide risk assessments
5 conducted by the EPA, the current risk assessment does not specifically attempt to assess
6 the risks of using adjuvants, unless specific information is available suggesting that the
7 risks may be substantial. For example, some adjuvants used in glyphosate formulations
8 may be as toxic as, and possibly more toxic than, glyphosate itself; accordingly, these
9 risks are addressed in the Forest Service risk assessment on glyphosate, which is not the
10 case with lambda-cyhalothrin.

11 **3.1.15. Impurities and Metabolites**

12 Lambda-cyhalothrin is extensively and rapidly metabolized in humans as well as
13 experimental mammals (U.S. EPA/HED 1997c). No information, however, has been
14 encountered on the risks associated with impurities in technical grade lambda-cyhalothrin
15 or metabolites of lambda-cyhalothrin. This situation is common in many pesticide risk
16 assessments. In general, pyrethroids are metabolized by hydrolysis, and these reactions
17 are catalyzed by mixed function oxidase. The hydrolysis of pyrethroids results in the
18 formation of more water soluble compounds that are likely to be less toxic and more
19 easily excreted than the parent compound (ATSDR 2003). U.S. EPA/OPP determined
20 that lambda-cyhalothrin, rather than the metabolites of lambda-cyhalothrin, is the agent
21 of toxicological concern (U.S. EPA/HED 1997c, p. 12).

22
23 Although there may be some basis for concern about impurities and metabolites, the
24 dose-response assessment for lambda-cyhalothrin is based on *in vivo* studies with
25 technical grade lambda-cyhalothrin. The underlying assumption in the current risk
26 assessment (as well as many other pesticide risk assessments) is that the toxicity of both
27 impurities and metabolites is encompassed by the use of *in vivo* studies that involve
28 exposures to both the impurities in technical grade lambda-cyhalothrin as well as the
29 metabolites of these compounds.

30
31 The assumption that the toxicity of metabolites is encompassed by the use of *in vivo*
32 studies in mammals is supported by the recent assessment of pyrethroids by ATSDR:

33
34 *Since the metabolites that have been identified in humans have*
35 *also been identified in other mammalian species, it is unlikely that*
36 *there are significant qualitative differences between humans and*
37 *other mammals in the major metabolic pathways for pyrethroids,*
38 *although some species differences do undoubtedly exist.*

39 ATSDR (2003, p. 91)

40 **3.1.16. Toxicological Interactions**

41 No information on mammals regarding the interactions of lambda-cyhalothrin with other
42 compounds was located in the available literature. As noted in the previous subsection,
43 detoxification by mixed-function oxidase is a common metabolic process for many
44 pyrethroids. Consequently, compounds that inhibit mixed-function oxidase may enhance

1 the toxicity of pyrethroids and compounds that induce mixed-function oxidase may
2 reduce the toxicity of pyrethroids. As discussed further in Section 4.1.2.4 (hazard
3 identification for terrestrial invertebrates), studies in honeybees demonstrate that
4 piperonyl butoxide and other inhibitors of mixed-function oxidase enhance the toxicity of
5 lambda-cyhalothrin by factors of up to about 100 (Johnson et al. 2006).
6

1 **3.2. EXPOSURE ASSESSMENT**

2 **3.2.1. Overview**

3 All exposure assessments for lambda-cyhalothrin are summarized in Worksheet E01 for
4 workers and in Worksheet E03 for the general public in the EXCEL workbook that
5 accompanies this risk assessment. All exposure assessments are based on six
6 applications separated at 2-week intervals at an application rate of 0.08 lb a.i./acre.

7
8 For workers applying lambda-cyhalothrin, two types of application methods are modeled:
9 airblast applications and high-pressure handwand applications. In non-accidental
10 scenarios involving the normal application of lambda-cyhalothrin, estimates of exposure
11 for workers are approximately 0.00003 (0.000002- 0.0001) mg/kg bw/day for airblast
12 applications and 0.0001 (0.000009-0.0006) mg/kg bw/day for high-pressure handwand
13 applications. All of the accidental exposure scenarios for workers involve dermal
14 exposures. Accidental exposures based on the assumption of first-order absorption lead
15 to very low estimates of absorbed doses and confidence in these estimates is high.
16 Accidental exposures involving zero-order dermal absorption lead to much higher
17 estimates of absorbed dose but confidence in these exposure assessments is very low.

18
19 For the general public (Worksheet E03), acute levels of exposures range from minuscule
20 (e.g., 1×10^{-7} mg/kg/day) to about 7 mg/kg bw. The upper bound of exposure of 7 mg/kg
21 bw is associated with the consumption of contaminated fish by subsistence populations
22 shortly after an accidental spill. This exposure scenario is highly arbitrary and does not
23 appear to be plausible for the Chico or Foresthill sites. The upper bound of the dose
24 associated with the consumption of contaminated vegetation—i.e., 0.17 mg/kg bw for the
25 Chico site—is a more plausible but still extreme exposure scenario. The other acute
26 exposure scenarios lead to much lower dose estimates. The lowest acute exposure levels
27 are associated with swimming in contaminated water.

28
29 The chronic or longer-term exposure levels are much lower than the estimates of
30 corresponding acute exposures. The highest longer-term exposure levels are associated
31 with the consumption of contaminated vegetation after a direct spray, and the upper
32 bound for this scenario is about 0.03 mg/kg/day. The lowest longer-term exposures are
33 associated with the consumption of contaminated surface water.

34 **3.2.2. Workers**

35 **3.2.2.1. General Exposures**

36 **3.2.2.1.1. Airblast Sprayer Applications**

37 As discussed in Section 2.3.1, applications at the Chico site involve airblast sprayer
38 applications within an enclosed cabin. These applications may be viewed as directed
39 foliar applications in that the spray nozzles are directed at that trees being treated. In
40 terms of worker exposure rates, however, these types of applications correspond most
41 closely to ground broadcast foliar applications.
42

1 No worker exposure studies are available involving ground broadcast applications of
2 lambda-cyhalothrin. Worker exposure rates for ground broadcast foliar
3 applications using in Forest Service risk assessments are typically taken as 0.0002
4 (0.00001- 0.0009) mg a.i./kg bw per lb a.i. applied. These rates are based on the study by
5 Nash et al. (1982) involving broadcast ground applications of 2,4-D, which is not directly
6 applicable to lambda-cyhalothrin because some of the equipment used to apply the 2,4-D
7 did not have cabs. Moreover, Nash et al. (1982) do not indicate whether the cabs were
8 enclosed. Applications of 2,4-D conducted in early 1980s most likely did not involve the
9 use of enclosed cabs.

10
11 The Pesticide Handlers Exposure Database (PHED Task Force 1995), which is used by
12 the U.S. EPA for deposition-based worker exposure assessments, provides some
13 information on deposition and inhalation exposures in workers involved in airblast
14 applications using both open and closed cabins. Based on the review of these data by
15 Keigwin (1998), dermal deposition in workers wearing gloves and single layer clothing is
16 0.24 mg/lb handled in open cabs and 0.019 mg/lb handled in closed cabs. Thus, the
17 worker protection factor for closed cabs, relative to open cabs, is about 0.92 [$1 - (0.019$
18 $\text{mg/lb handled} \div 0.24 \text{ mg/lb handled})$]. Keigwin (1998) also notes that the inhalation
19 exposure rate is 0.0045 mg/lb handled in open cabs and 0.00045 mg/lb handled in closed
20 cabs., which leads to a protection factor for closed cabs, relative to open cabs of 0.9 [$1 -$
21 $(0.00045 \text{ mg/lb handled} \div 0.0045 \text{ mg/lb handled})$].

22
23 For the current Forest Service risk assessment, either the PHED exposure estimates
24 (modified to account for dermal exposure) or the standard Forest Service exposure rates
25 (modified to account for enclosed cabs) could be used.

26
27 Based on a worker protection factor of 0.9 and the standard worker exposure rates of
28 0.0002 (0.00001-0.0009) mg a.i./kg bw per lb a.i. applied used in Forest Service risk
29 assessments, the functional worker exposure rates would be 0.00002 (0.0000001-
30 0.00009) mg/kg bw per lb a.i. handled.

31
32 Based on dermal absorption rates of 0.0012 (0.00041- 0.01) day⁻¹ (Section 3.1.3.2), the
33 dermal deposition rate of 0.19 mg/lb handled from Keigwin (1998) corresponds to
34 absorbed dermal doses of about 0.00023 (0.000078- 0.0019) mg/lb handled. Adding the
35 inhalation exposure rate of 0.00045 mg/lb handled and assuming 100% inhalation
36 absorption (a standard approach used in EPA exposure assessments), the total absorbed
37 dose would be 0.00068 (0.00053- 0.00024) mg/lb handled. The EPA uses body weights
38 of 60 kg for female workers and 70 kg for male workers. Using the lower body weight of
39 60 kg, which results in higher estimates of mg/kg bw doses, the worker exposure rates
40 would be 0.000011 (0.0000088-0.000039) mg/kg bw per lb a.i. handled.

41
42 The estimates based on the standard Forest Service exposure rate are higher than those
43 using the PHED approach by a factor of about 1.8, based on the central estimates, and a
44 factor of about 2.3, based on the upper bound estimate. Thus, the two different methods
45 of estimating worker exposures yield reasonably similar results. The lower bound of the
46 PHED estimate is a factor of about 87 higher than the lower bound of the estimate based

1 on the Forest Service method; however, the lower bounds have little impact on the
2 assessment of risk.

3
4 The current Forest Service risk assessment uses the worker exposure rates normally used
5 in Forest Service risk assessments with a worker protection factor of 0.9, as discussed
6 above. This approach is a modestly more conservative than using the estimates derived
7 from PHED. As discussed further in Section 3.4.2 (risk characterization for workers), the
8 HQs for non-accidental exposures in workers are substantially below the level of
9 concern. Thus, the use of the somewhat more conservative exposure assumptions does
10 not have an impact on the qualitative interpretation of the risk characterization.

11
12 In estimating the absorbed dose rate, the amount of pesticide handled must be estimated.
13 This amount is estimated as the product of the application rate (lb a.i./acre) and the
14 number of acres treated. Standard Forest Service risk assessments typically assume that
15 ground spray workers may treat up to 21 acres/hour over a 6- to 8-hour period, which is
16 not the case for the pesticide applications made at the Chico site. As discussed in Section
17 2, the treated area consists of only 84 acres, and applications are made by five workers—
18 i.e., five sets of spray vehicles—over about a 5-hour period. Thus, each worker would
19 treat about 17 acres. Consequently, Worksheet C01 in the EXCEL workbook for the
20 Chico site (Attachment 1 to the current Forest Service risk assessment) is modified to
21 specify an application period of 4 hours and a treatment rate of 4.25 acres/hour—i.e., a
22 treatment area of 17 acres per worker.

23 ***3.2.2.1.2. High Pressure Foliar Applications***

24 Applications at the Foresthill site are made to individual trees using high-pressure nozzles
25 (Section 2.3). These types of applications are similar to bark applications covered in the
26 recent Forest Service risk assessment on carbaryl (SERA 2009b). As discussed in the
27 carbaryl risk assessment, worker exposure rates were taken as 0.003 (0.0003-0.01) mg/kg
28 bw per lb handled. These rates are based on numerous studies involving estimated
29 absorption rates for backpack applications of various pesticides (SERA 2007a).

30
31 No studies are available on high-pressure nozzle applications or standard backpack
32 applications of lambda-cyhalothrin. Chester et al. (1992), however, estimated absorbed
33 doses of lambda-cyhalothrin (based on urinary excretion of metabolites) after indoor
34 applications using hand-held compression sprayers. Average absorbed doses in workers
35 were estimated at 54 µg/day with an approximate range of about 9 to 80 µg/day, using a
36 typical worker body weight of 60 kg. Thus, the mg/kg bw absorbed doses would be
37 about 0.0009 (0.00015-0.0013) mg/kg bw. The Chester et al. (1992) study further
38 indicates that the average amount handled per worker was about 0.065 kg, which
39 corresponds to about 0.14 lb [0.065 kg x 2.205 lb/kg = 0.143325 lb]. Thus, the worker
40 exposure rates are estimated at 0.006 (0.001-0.009) mg/kg bw per lb handled [0.0009
41 (0.00015-0.0013) mg/kg bw ÷ 0.14 lb]. The central estimate of 0.006 mg/kg bw per lb
42 handled is a factor of two higher than the standard backpack rate of 0.003 mg/kg bw per
43 lb handled. Given the variability in both sets of values—i.e., a factor of about 33 in the
44 standard Forest Service estimates and a factor of 9 in the Chester study—the differences
45 in the central estimates are insubstantial.

1
2 For the current Forest Service risk assessment, the standard backpack exposure rates of
3 0.003 (0.0003-0.01) mg/kg bw per lb handled are used to estimate worker exposures at
4 the Foresthill site. These exposure rates, however, do not consider or incorporate the use
5 of worker protective equipment. As discussed in Section 2.3, applicators will wear
6 waterproof Tyvek overalls with a built-in hood, full face shield, protective glasses, rubber
7 boots, and chemical gloves taped into the coveralls. As noted in the analysis of PHED
8 data by Keigwin (1998), a protection factor of 0.9 is typical for the use of chemical
9 resistant gloves, which is identical to the protection factor noted by Nigg (1998) for the
10 use of protective clothing in many pesticide applications. For lambda-cyhalothrin,
11 Hughes et al. (2005) report single layer clothing penetration factors of 5.3-7.2%. Thus, it
12 appears that the use of a general worker protection factor of 0.9 is appropriate and
13 probably conservative—i.e., the use of a protection factor of 0.9 may overestimate
14 exposures for workers wearing Tyvek coveralls.

15
16 In addition to the worker exposure rate and clothing protection factor, worker exposure
17 level depends on the amount of material handled. Information from the Foresthill site
18 indicates that an individual worker treats about 150 trees per day at application rates of
19 0.002-0.004 lbs a.i./tree. Based on 0.003 lbs a.i./tree as a central estimate, the amount of
20 lambda-cyhalothrin handled per day is 0.45 (0.3-0.6) lbs [0.003 (0.002-0.004) lbs a.i./tree
21 x 150 trees]. A custom modification is made to Worksheet C01 of the EXCEL workbook
22 for the Foresthill site (Attachment 2 to the current Forest Service risk assessment) to use
23 these values in calculating the absorbed dose rates for workers.

24 **3.2.2.2. Accidental Exposures**

25 Accidental exposures are most likely to involve splashing a solution of the pesticide into
26 the eyes or contaminating the surface of the skin. Quantitative exposure scenarios for eye
27 exposures are not developed in this or other Forest Service risk assessments. As
28 discussed in Section 3.1.11 (Irritation and Sensitization), lambda-cyhalothrin is classified
29 as a Category II eye irritant and splashing a solution of lambda-cyhalothrin into the eyes
30 would presumably cause eye irritation. This effect is considered qualitatively in the risk
31 characterization for workers (Section 3.4.2).

32
33 Accidental dermal exposure to lambda-cyhalothrin is considered quantitatively in this
34 risk assessment. The two types of dermal exposure that are modeled include direct
35 contact with a pesticide solution and accidental spills of the pesticide onto the surface of
36 the skin. Furthermore, two exposure scenarios are developed for each of the two types of
37 dermal exposure, and the estimated absorbed dose for each scenario is expressed in units
38 of mg chemical/kg body weight. Both sets of exposure scenarios are summarized in
39 Worksheet E01, which references other worksheets in which the calculations are
40 specified.

41
42 Exposure scenarios involving direct contact with solutions of pesticides are characterized
43 either by immersion of the hands in a field solution for 1 minute or wearing pesticide
44 contaminated gloves for 1 hour. For these exposure scenarios, the key assumption is that
45 wearing gloves grossly contaminated with a chemical solution is equivalent to immersing

1 the hands in the solution. In both cases, the chemical concentration in contact with the
2 skin and the resulting dermal absorption rate are essentially constant. These are standard
3 exposure scenarios used in all Forest Service risk assessments. For the current risk
4 assessment of lambda-cyhalothrin, however, these exposure scenarios are improbable.
5 As discussed in Section 2.3, applications at the Chico site involve workers in enclosed
6 cabins. Applications at the Foresthill site involve the use of chemical resistant gloves
7 that are taped to Tyvek overalls. Thus, it is not likely that either of these exposure
8 scenarios would occur during applications of lambda-cyhalothrin.

9
10 Another issue with exposure scenarios involving hand immersion and contaminated
11 gloves involves the estimates of the dermal absorption rates. For both scenarios (hand
12 immersion and contaminated gloves), the assumption of zero-order absorption kinetics is
13 appropriate. For these types of exposures, the rate of absorption is estimated, based on a
14 zero-order dermal absorption rate (K_p). As discussed in Section 3.1.3.2, the K_p for
15 lambda-cyhalothrin is estimated using a QSAR algorithm developed by the EPA;
16 however, confidence in these estimates for lambda-cyhalothrin is very low, as discussed
17 further in the risk characterization for workers (Section 3.4.2).

18
19 Exposure scenarios involving chemical spills onto the skin are characterized by a spill on
20 to the lower legs as well as a spill on to the hands and are based on the assumption that a
21 certain amount of the chemical adheres to the skin. These types of accidents would most
22 probably occur during the mixing/loading process. The absorbed dose is calculated as the
23 product of the amount of chemical on the surface of the skin (i.e., the amount of liquid
24 per unit surface area multiplied by the surface area of the skin over which the spill occurs
25 and the chemical concentration in the liquid), the first-order absorption rate, and the
26 duration of exposure. As with the zero-order dermal absorption rate, the first-order
27 absorption rate (k_a) is derived in Section 3.1.3.2. This absorption rate is based on a
28 human study reviewed and accepted by the U.S. EPA, and confidence in these first-order
29 dermal absorption rates is much higher than confidence in the estimates of K_p .

30 **3.2.3. General Public**

31 **3.2.3.1. General Considerations**

32 **3.2.3.1.1. Likelihood of Exposure**

33 The likelihood that members of the general public will be exposed to pesticides in Forest
34 Service applications is highly variable. In some Forest Service applications, pesticides
35 may be applied in recreational areas, including campgrounds, picnic areas, and trails. In
36 other instances, pesticides may be applied in relatively remote areas and the probability
37 that members of the general public will be exposed to the pesticides is remote. As
38 discussed in Section 2.6, the current risk assessment on lambda-cyhalothrin is somewhat
39 atypical in that two specific sites are considered, the Chico site and the Foresthill site.
40 The Chico site is in a populated area, and the site is used by general public. Thus, the
41 probability of exposure to pesticides applied at the site is high. The Foresthill site, on the
42 other hand, is in a relatively remote area. While the general public is not excluded, the
43 Foresthill site is not designed for recreational activities, and the probability of exposure is
44 relatively low.

1 **3.2.3.1.2. Summary of Assessments**

2 The exposure scenarios developed for the general public are summarized in Worksheet
3 E03 of the EXCEL workbooks that accompany this risk assessment. As with the worker
4 exposure scenarios, details about the assumptions and calculations used in these
5 assessments are given in the worksheets that accompany this risk assessment
6 (Worksheets D01–D11).

7
8 As summarized in Worksheet E03, the kinds of exposure scenarios developed for the
9 general public include acute accidental, acute non-accidental, and longer-term or chronic
10 exposures. The accidental exposure scenarios assume that an individual is exposed to the
11 compound of concern either during or shortly after its application. What is more, the
12 nature of the accidental exposures is intentionally extreme. Non-accidental exposures
13 involve dermal contact with contaminated vegetation as well as the consumption of
14 contaminated fruit, vegetation, water, and fish. The longer-term or chronic exposure
15 scenarios parallel the acute exposure scenarios for the consumption of contaminated fruit,
16 water, and fish. All of the non-accidental exposure scenarios are based on levels of
17 exposure to be expected in the routine uses of lambda-cyhalothrin. Nonetheless, the
18 upper bounds of the exposure estimates for the non-accidental scenarios involve
19 conservative assumptions intended to reflect exposure for the MEI (*Most Exposed*
20 *Individual*).

21
22 While the current risk assessment is focused on two specific sites, the exposure
23 assessments considered in this risk assessment are generally similar to those used in all
24 Forest Service risk assessments. This approach is taken because the Forest Service uses
25 lambda-cyhalothrin at locations other than Chico and Foresthill. For example, lambda-
26 cyhalothrin has been considered in a risk assessment for Forest Service Region 8
27 (Mistretta 2007). Thus, including all of the standard exposure scenarios will enhance the
28 usefulness of the current risk assessment in other Forest Service programs.

29
30 Not all of the standard exposure scenarios, however, are applicable to the Chico and
31 Foresthill sites. For example, all Forest Service risk assessments consider the direct
32 spray of and pesticide drift to a generic pond and a generic stream and separately
33 consider pesticide transport to generic ponds and streams using Gleams-Driver. For the
34 current Forest Service risk assessment on lambda-cyhalothrin, however, there is sufficient
35 information on the Chico and Foresthill sites to consider both drift and pesticide transport
36 in Gleams-Driver simulations. Thus, while the direct spray scenarios are discussed in
37 detail in the current Forest Service risk assessment, they are not considered in the risk
38 characterization for the Chico and Foresthill sites. Other examples in which the exposure
39 scenarios are either not relevant to the Chico or Foresthill sites or have been modified to
40 reflect conditions at the Chico and Foresthill sites are noted in the following subsections
41 and discussed further in the risk characterization.

42 **3.2.3.2. Direct Spray**

43 Direct sprays involving ground applications are modeled similarly to accidental spills for
44 workers (Section 3.2.2.2). In other words, the scenarios assume that an individual is
45 sprayed with a chemical solution, some of which remains on the skin and is absorbed by

1 first-order kinetics. Two direct spray scenarios are included in this risk assessment: one
2 for a young child (D01a) and the other for a young woman (D01b).

3
4 The exposure scenario involving the young child assumes that a naked child is sprayed
5 directly with a chemical during a ground broadcast application and is completely covered
6 (i.e., 100% of the surface area of the body is exposed). This exposure scenario is
7 intentionally extreme and is used to assess worst-case dermal exposure levels.

8
9 The exposure scenario involving the young woman (Worksheet D01b) is somewhat less
10 extreme and more plausible. In this scenario, it is assumed that the woman is
11 accidentally sprayed over the feet and lower legs. The preference for using a young
12 woman rather than an adult male in many of the exposure assessments relates to concerns
13 for both the *Most Exposed Individual* (MEI) as well as the most sensitive individual.
14 Based on general allometric considerations, the smaller the individual, the greater will be
15 the chemical doses per unit body weight (e.g., Boxenbaum and D'Souza. 1990).
16 According to standard reference values used in exposure assessments (e.g., U.S.
17 EPA/ORD 1989), the female body size is smaller than that of males. Thus, in direct
18 spray exposure scenarios, females are subject to somewhat higher doses than males.
19 More significantly, reproductive effects are a major concern in all Forest Service risk
20 assessments. Consequently, exposure levels for a young woman of reproductive age are
21 used in order to better assess the potential for adverse effects in the population at risk
22 from potential reproductive effects—i.e., the most exposed and the most sensitive
23 individual.

24
25 For this exposure scenario, assumptions are made regarding the surface area of the skin
26 and the body weight of the individual, as detailed in Worksheet A03. The rationale for
27 and sources of the specific values used in these and other exposure scenarios is provided
28 in the documentation for the worksheets (SERA 2009a) as well as the documentation for
29 the preparation of Forest Service risk assessments (SERA 2007a).

30
31 No modifications to these exposure scenarios as well as other accidental exposure
32 scenarios are made to reflect conditions at the Chico or Foresthill sites. By definition,
33 accidental exposures are unanticipated; however, they must be considered in order to
34 assess the consequences of improbable exposure levels due to mischance.

35 ***3.2.3.3. Dermal Exposure from Contaminated Vegetation***

36 The exposure scenario involving contaminated vegetation assumes that the herbicide is
37 sprayed at a given application rate and that a young woman comes in contact with the
38 sprayed vegetation or with other contaminated surfaces sometime after the spray
39 operation (D02). This exposure scenario depends on estimates of dislodgeable residue (a
40 measure of the amount of the chemical that could be released from the vegetation) and
41 the availability of dermal transfer rates (i.e., the rate at which the chemical is transferred
42 from the contaminated vegetation to the surface of the skin).

43
44 Most Forest Service risk assessments use a default dislodgeable residue rate of 0.1 of the
45 application rate, based on a field simulation study which measured dermal exposure

1 levels in humans after an application of 2,4-D (Harris and Solomon 1992). For lambda-
2 cyhalothrin, Estes and Buck (1990) assayed substantially higher initial dislodgeable
3 residues in applications to cotton. In this study, lambda-cyhalothrin was applied at a rate
4 of 0.028 kg/ha, equivalent to 0.28 $\mu\text{g}/\text{cm}^2$ [$28,000 \text{ mg}/10,000 \text{ m}^2 = 2.8 \text{ mg}/\text{m}^2 = 2800$
5 $\mu\text{g}/10,000 \text{ cm}^2 = 0.28 \mu\text{g}/\text{cm}^2$], and initial dislodgeable residues were measured at 0.1
6 $\mu\text{g}/\text{cm}^2$. Thus, the dislodgeable residue rate is about 0.36 of the nominal application rate
7 [$0.1 \mu\text{g}/\text{cm}^2 / 0.28 \mu\text{g}/\text{cm}^2 = 0.357$]. In the absence of information on dislodgeable
8 residues for lambda-cyhalothrin in turf, the dislodgeable residue rate of 0.36 from the
9 study by Estes and Buck (1990) is used in the current Forest Service risk assessment.

10
11 This exposure scenario is plausible for the Chico site but less plausible for the Foresthill
12 site. Even at the Chico site, this exposure scenario probably overestimates exposures to
13 members of the general public because the scenario assumes that the individual is on the
14 spray site very soon after the pesticide is applied. At the Chico site, however, members
15 of the general public are not likely to be at the spray site during the application period
16 (which occurs only in the evening or at night). Therefore, dermal contact with
17 contaminated vegetation would more likely occur downwind from the application site
18 several hours after the applications are made. As discussed further in the risk
19 characterization, the HQs associated with this very conservative exposure scenario are far
20 below the level of concern. Thus, no refinements to this exposure scenario are made.

21 **3.2.3.4. Contaminated Water**

22 **3.2.3.4.1. Accidental Spill**

23 The accidental spill scenario assumes that a young child consumes contaminated water
24 shortly after an accidental spill of a field solution into a small pond. The specifics of this
25 scenario are given in Worksheet D05. Because this scenario is based on the assumption
26 that exposure occurs shortly after the spill, no dissipation or degradation is considered.
27 Since this exposure scenario is based on assumptions that are somewhat arbitrary and
28 highly variable, it may overestimate exposure. The actual chemical concentrations in the
29 water will vary according to the amount of compound spilled, the size of the water body
30 into which it is spilled, the time at which water consumption occurs relative to the time of
31 the spill, and the amount of contaminated water consumption. To reflect the variability
32 inherent in this exposure scenario, a spill volume of 100 gallons (in the range of 20-200
33 gallons) is used to reflect plausible spill events. The lambda-cyhalothrin concentrations
34 in the field solution are also varied to reflect the plausible range of concentrations in field
35 solutions—i.e., the material that might be spilled—using the same values as in the
36 accidental exposure scenarios for workers (Section 3.2.2.2). Based on these assumptions,
37 the estimated concentration of lambda-cyhalothrin in a small pond ranges from about
38 0.007 to 0.14 mg/L, with a central estimate of about 0.04 mg/L (Worksheet D05).

39
40 These exposure scenarios are not directly relevant to either the Chico or Foresthill sites.
41 As discussed further in Section 3.2.3.4.3 (Gleams-Driver Modeling), these exposure
42 scenarios are probably irrelevant for the Foresthill site. Consistent with observations
43 from the Forest Service (USDA/FS 1997), the Gleams-Driver modeling suggests that
44 McBride Creek will be dry during periods when lambda-cyhalothrin is applied.
45 Similarly, runoff discharge to Comanche Creek is not likely during periods in which

1 lambda-cyhalothrin will be applied. Nonetheless, the Forest Service has indicated that
2 pools of water will exist along Comanche Creek during periods when lambda-cyhalothrin
3 is applied (USDA/FS 1998). In the event of a severe spill, contamination of surface
4 water is plausible. Nonetheless, Comanche Creek is in an area where there is relatively
5 easy access to drinking water. The consumption of water from stagnant sections of
6 Comanche Creek seems implausible.

7 **3.2.3.4.2. Accidental Direct Spray/drift for a Pond or Stream**

8 The exposure scenarios involving drift are less severe but more plausible than the
9 accidental spill scenario described above. U.S. EPA typically uses a 2-meter-deep pond
10 to develop exposure assessments (SERA 2007b). If such a pond is directly sprayed with
11 lambda-cyhalothrin at an application rate of 0.08 lb a.i./acre, the peak concentration in
12 the pond would be about 0.009 mg/L, equivalent to 9 µg/L or 9 ppb (Worksheet D10a).
13 Worksheet D10a also models concentrations at distances of 25-900 feet down wind,
14 based on standard values adapted from AgDrift (SERA 2009a). Based on these
15 estimates, lambda-cyhalothrin concentrations in a small pond contaminated by drift
16 would range from about 0.000003 mg/L (3 part per trillion) to 0.00007 mg/L (70 parts
17 per billion).
18

19 Similar calculations can be made for the direct spray of or drift into a stream. For this
20 scenario, the resulting water concentrations depend on the surface area of the stream and
21 the rate of water flow in the stream. The stream is assumed to be about 6 feet wide
22 (1.82 meters) and it is assumed that the pesticide is applied along a 1038-foot (316.38
23 meters) length of the stream with a flow rate of 710,000 L/day. Based on these values,
24 the concentration in stream water after a direct spray at an application rate of 0.08 lb
25 a.i./acre is estimated at about 0.007 mg/L (7 parts per billion). Much lower
26 concentrations, ranging from about 0.000002 mg/L (2 part per trillion) to 0.00006 mg/L
27 (60 part per trillion) are estimated based on drift at distances of 25-900 feet (Worksheet
28 D10b).
29

30 As noted in the previous subsection, Comanche Creek (Foresthill) is not likely to have
31 any flow during periods when lambda-cyhalothrin is applied. Thus, the drift scenarios do
32 not appear to be relevant for the Foresthill site.

33 **3.2.3.4.3. GLEAMS Modeling**

34 Forest Service risk assessments use Gleams-Driver to estimate expected peak and longer-
35 term pesticide concentrations in surface water. Gleams-Driver serves as a preprocessor
36 and postprocessor for GLEAMS, a field scale model developed by the USDA/ARS and a
37 program used for many years in Forest Service and other USDA risk assessments
38 (SERA 2007b). In typical Forest Service risk assessments, 27 different Gleams-Driver
39 simulations are used to model nine generic sites representing combinations of
40 temperature (cool, moderate, and warm) and rainfall (sparse, moderate, and heavy), and
41 each site is modeled for three types of soil (clay, loam, and sand) to represent different
42 runoff potentials (high, moderate, and low). The generic Gleams-Driver runs do not
43 typically consider drift because drift is addressed in other worksheets as discussed in
44 Section 3.2.3.4.2.

1
2 In order to make this risk assessment more generally useful across Forest Service regions,
3 two sets of these standard Gleams-Driver simulations were conducted, and the results are
4 detailed in Appendix 6 (a single application) and Appendix 7 (six applications at 2-week
5 intervals). Because the current risk assessment is focused on two specific sites, Chico
6 and Foresthill, these generic simulations are not used directly in the current risk
7 assessment but are discussed briefly in Section 3.2.3.4.3.3.

8 **3.2.3.4.3.1. Methods for Site-Specific Modeling**

9 Table 4 summarizes the site-specific input parameters used in the Gleams-Driver
10 simulations. The sources for this information, as detailed in the footnotes to Table 4, are
11 taken from standard sources such as the USDA Soil Survey and the USGS National
12 Water Information System. Details in the use of these information sources are provided
13 in the documentation for Gleams-Driver (SERA 2007b). In cases where some site-
14 specific input values for soils were not available, default values were taken from Knisel
15 and Davis (2000) based on the soil types specified in the USDA Soil Survey. Other
16 information on the sites was provided by the Forest Service. Table 5 summarizes the
17 chemical-specific values used in the Gleams-Driver simulations. For the most part, these
18 values are taken from the GLEAMS documentation (Knissel and Davis 2000) and other
19 standard sources (Tomlin 2005; USDA/ARS 1995).

20
21 Climate files for the Chico and Foresthill sites were generated using Cligen Version 5.2
22 as detailed in the documentation for Gleams-Driver (SERA 2007b, Section 6.1.1). For
23 the Chico site, the weather file was generated based on data from the Chico Experiment
24 Station. Cligen does not include weather data from Foresthill, California, the town
25 nearest to the Foresthill Genetics Center. Consequently, the linear interpolation
26 algorithm in Cligen was used to generate the climate file for the Foresthill site. To assess
27 the reliability of the interpolation algorithm for Foresthill, average monthly precipitation
28 data for Foresthill, California over the period from 2007 to 2009 were obtained from
29 www.foresthillweather.com and compared with the monthly average precipitation
30 simulated by Cligen. As illustrated in Figure 4, the Cligen simulations reasonably
31 approximate the rainfall patterns for Foresthill, California.

32
33 As is typical for Gleams-Driver runs, 100 simulations were conducted for each site. As
34 necessary, the Gleams-Driver simulations were modified for the Chico and Foresthill
35 sites. As noted above, typical Forest Service risk assessments do not consider drift in
36 Gleams-Driver simulations and all simulations are conducted at an application rate of 1 lb
37 a.i./acre. For the site-specific Gleams-Driver runs, the simulations did consider drift and
38 all simulations were conducted at the anticipated application rate of 0.08 lb a.i./acre with
39 six applications based at 2-week intervals beginning in May. Consequently, Worksheet
40 B04 in each of the EXCEL workbooks that accompanies this risk assessment was
41 modified to accept the modeling results and convert the results to water contamination
42 rates (WCR) in units of mg/L per lb a.i. applied per acre. Note, however, that all risks are
43 assessed at the application rates specified by the Forest Service.

44

1 As discussed in Section 2.3.3, lambda-cyhalothrin is not currently used at either the
2 Chico or Foresthill sites and the application rate, number of applications, and application
3 interval that might be used at these sites have not yet been determined. Consequently, a
4 separate series of Gleams-Driver simulations were conducted for three applications with
5 a four week application interval using an application rate of 0.16 lb a.i./acre. The
6 application rate of 0.16 lb a.i./acre is selected because this is the recommended
7 application rate for the control of coneworms or seed bugs. The application interval was
8 set to four weeks so that the period over which applications would be made would
9 correspond to the six application scenario discussed above that uses a two week
10 application interval.

11
12 Drift estimates were made using AgDrift Version 2.0.05 based on information provided
13 by the Forest Service. For airblast applications at the Chico site, the Forest Service has
14 indicated that applications may be made as close as 50 feet to the stream. AgDrift
15 provides five options or sub-scenarios for airblast drift: *normal*, *dense*, *sparse*, *vineyard*,
16 and *orchard*. In standard Forest Service risk assessments, drift estimates are based on the
17 *normal* option for airblast applications for which the drift at 50 feet downwind is
18 approximately 0.001 of the application rate. The *orchard* option in AgDrift, however,
19 yields substantially higher estimates of drift—i.e., about 0.02. For the Gleams-Driver
20 modeling, the higher drift estimate of 0.02 is used. As discussed in the following
21 subsection, this does not have a substantial impact on the peak and longer-term estimates
22 of lambda-cyhalothrin in water.

23
24 The other factor that impacts the amount of pesticide reaching the stream involves the
25 length of the stream that might be impacted as well as the stream width. The stream
26 width is taken as 15 feet, based on information from the Forest Service (Bakke 2009c).
27 The stream length that might be impacted by drift is very difficult to estimate with any
28 precision and would likely depend on wind direction as well as the geometry of the
29 stream and drift cloud in the impact area. For the Gleams-Driver simulations, the length
30 of the stream impacted by drift is taken as 500 feet, approximately one quarter of the
31 longest flow path in the field.

32
33 For the Foresthill site, the approach taken to estimating drift is similar to that used for the
34 Chico site. As discussed in the documentation for Gleams-Driver, AgDrift and other drift
35 modeling programs do not provide drift estimates for handwand directed foliar
36 applications. Thus, for the Foresthill site, drift was estimated using 50 percentile rates for
37 course droplets in ground boom applications. This is the same approach used in Forest
38 Service risk assessments for backpack applications (SERA 2009a). As noted in Table 4,
39 the drift is estimated to range from 0.0002 to 0.002, using a uniform distribution.

40
41 As indicated in Table 4, both Comanche Creek (Chico site) and McBride Creek
42 (Foresthill site) are ephemeral. Typically, water will not be actively flowing in these
43 creeks during the summer months (USDA/FS 1997,1998). The only additional
44 information on stream flow comes from USDA/FS (1998), which indicates that the
45 maximum flow rate for Comanche Creek is about 150 cubic feet per second. No

1 information on these two creeks is available at the USGS website for surface water
2 statistics (<http://waterdata.usgs.gov/nwis>).

3
4 Gleams-Driver Version 1.8 does not accommodate ephemeral streams. In generic
5 Gleams-Driver simulations, minimal stream flow is set at 710,000 L/day or about 0.3
6 cubic feet per second. This minimum flow rate is used in the simulations for both the
7 Chico and Foresthill sites; however, it has no impact on the estimated concentrations of
8 lambda-cyhalothrin at the Foresthill site, because concentrations modeled during days in
9 which no runoff occurs are essentially zero. At the Chico site, the minimum flow rate
10 does have an impact on the estimated concentrations for days on which applications are
11 made, because drift is significant at the Chico site. Both of these factors are discussed
12 further in Section 3.2.3.4.3.2.

13
14 It should also be noted that the use of a minimum flow rate is not a conservative
15 assumption in the sense that pesticide concentrations in streams will be overestimated.
16 The pesticide will be transported to surface water only on days in which rainfall causes
17 runoff and/or percolation. The pesticide will enter the water at the concentration of the
18 pesticide in runoff or percolate and will be diluted by the minimal flow rate. Minimum
19 flow is intended to represent reasonable upstream contributions to stream flow on days in
20 which significant rainfall occurs.

21
22 Another modification made for the simulations at Chico and Foresthill involves sediment
23 binding. In the development of Gleams-Driver, sediment binding was incorporated into
24 the pond model but not into the stream model. The decision not to model sediment
25 binding in streams was taken as a matter of convenience and because most pesticides
26 assessed at the time that Gleams-Driver was developed have high water solubilities and
27 low K_{oc} values. Thus, sediment binding is not a major consideration for these pesticides.

28
29 For lambda-cyhalothrin, sediment binding cannot be ignored. As indicated in Table 5,
30 the water solubility of lambda-cyhalothrin is low (i.e., 0.005 mg/L) and the K_{oc} for
31 lambda-cyhalothrin is high (i.e., 180,000 mL/g). More specifically, Hadfeld et al. (1993)
32 demonstrated that hydrosol acts a significant reservoir for lambda-cyhalothrin, which
33 will also be the case for other highly lipophilic compounds.

34
35 To consider sediment binding, the source code for Gleams-Driver Version 1.8 was
36 modified to consider sediment binding in streams using an algorithm analogous to that
37 used in ponds (SERA 2007b, Section 7.4.1, p. 58). This change to Gleams-Driver will be
38 incorporated into the next public release of Gleams-Driver.

39
40 In addition to sediment binding, investigators have demonstrated that macrophytes can
41 rapidly absorb and degrade lambda-cyhalothrin (Armitage et al. 2008; Hand et al. 2001;
42 Leistra et al. 2004). Because the two streams under consideration at the Chico and
43 Foresthill sites are ephemeral, it does not seem likely that macrophyte populations will be
44 dense. In addition, quantitative methods for considering macrophyte uptake and
45 degradation are not implemented in Gleams-Driver, and to implement them is beyond the
46 scope of the current analysis. Thus, no attempt was made to consider the impact of

1 macrophytes quantitatively in the current modeling.

2
3 A major factor in the significance of sediment binding to concentrations in streams or
4 ponds involves the depth of the water column and the depth of the sediment that is
5 functionally involved in sediment binding. In generic Gleams-Driver pond simulations,
6 the assumption is made that the water depth is 2 meters and the sediment depth is 2 cm.
7 Thus, the ratio of the sediment depth to the depth of the water column is 0.01 [0.02 m ÷
8 2m]. The actual sediment depth that is important for binding in streams or ponds is likely
9 to vary substantially among sites. Without conducting a site-specific calibration study to
10 determine the sediment depth in a stream, which is important for binding, no objective
11 method for setting the ratio of the sediment depth to water column is apparent. For the
12 current modeling using Gleams-Driver, the ratio of 0.01 is used.

13
14 According to Liu et al. (2004), copper may increase the rate of lambda-cyhalothrin
15 photolysis in water by a factor of about 50%. No information is available on the levels of
16 copper in the two streams under consideration; furthermore, it is not clear that this type of
17 information could be effectively incorporated into the current modeling.

18
19 A final consideration in the estimates of concentrations of lambda-cyhalothrin in surface
20 water involves buffers. As summarized in Table 4, there is a 50-foot buffer between the
21 stream and the application area at the Chico site and a 200-foot buffer at the Foresthill
22 site. Neither GLEAMS nor PRZM (the field scale vadose zone model used by EPA)
23 explicitly consider buffers. The quantitative consideration of buffers is very complex,
24 and buffer efficiency at a specific site can be influenced by many factors other than
25 distance —e.g., macropore flow and surface characteristics. Nonetheless, the Gleams-
26 Driver modeling does use information on both the treated area and the total drainage area
27 for the streams at the two sites. This consideration partially and conservatively reflects
28 the potential impact of the buffer zones at the two sites.

29 **3.2.3.4.3.2. Results of Site-Specific Modeling**

30 The Gleams-Driver simulations for the streams at the Chico and Foresthill sites are
31 summarized in Table 6. The central estimates are given as the median values for the 100
32 simulations at each site, and the upper and lower bounds are taken as the empirical 0.025
33 and 0.975 percentiles—i.e., empirical 95% confidence limits.

34
35 As summarized in Table 6, concentrations in the streams at the Foresthill and Chico sites
36 are similar. Taking the results for the six applications at a rate of 0.08 lb a.i./acre, the
37 central estimate of the peak concentrations are a factor of 1.4 higher at the Foresthill site,
38 relative to the Chico site—i.e., 7.73×10^{-6} (Foresthill) ÷ 5.35×10^{-6} (Chico) ≈ 1.4449).
39 Conversely, the central estimate of the longer concentrations are a factor of about 1.9
40 higher at the Chico site, relative to the Foresthill site—i.e., 1.56×10^{-7} (Chico) ÷ $8.02 \times$
41 10^{-8} (Foresthill) ≈ 1.9451). Given the broad range of modeled concentrations at each site,
42 these differences are insubstantial. The general similarities in concentrations between the
43 two sites appear to reflect offsetting differences between the sizes of the treated areas, the
44 total drainage areas, the representative slopes, and soil characteristics.

45

1 In addition to the scenarios for six applications of 0.08 lb a.i./acre using a two week
2 interval, Table 6 also summarizes the results of Gleams-Driver simulations at the Chico
3 and Foresthill sites for three applications of 0.16 lb a.i./acre using a 4-week application
4 interval. The differences in estimated concentrations in the streams compared to the six-
5 application scenario are not remarkable. This is to be expected given that both the 0.16 lb
6 a.i./acre and 0.08 lb a.i./acre simulations use the same weather and soil data and both
7 simulations involve the same cumulative application over the same application period.

8 **3.2.3.4.3.3. Results of Generic Modeling**

9 Results of the generic modeling from Gleams-Driver are presented in Appendix 6 (a
10 single application) and Appendix 7 (six applications at 2-week intervals). Unlike the
11 simulations for the Chico and Foresthill sites, discussed in the previous subsection, all of
12 the generic runs were conducted at a unit application rate of 1 lb a.i./acre. As a
13 convention, the workbooks created by WorksheetMaker are designed to accommodate a
14 unit application rate of 1 lb/acre, referred to as a water contamination rate (WCR).
15 Expected concentrations are derived from this rate based on the actual application rate
16 that is used in the workbook. All of the concentrations from the generic modeling can be
17 viewed as water contamination rates (WCRs)—i.e., the concentrations anticipated in
18 water at an application rate of 1 lb a.i./acre per application. In other words, Appendix 7
19 models six applications, each at 1 lb a.i./acre, with a 2-week interval between each
20 application. These generic runs are not used quantitatively in the current risk assessment.
21 The results for the generic modeling of the stream, however, are used in the database for
22 WorksheetMaker.

23
24 The generic runs for the stream are the only values that are directly comparable to the
25 site-specific runs discussed in the previous subsection. Notably, the upper bounds of the
26 WCRs from the generic runs are higher than the upper bounds of the WCRs derived from
27 the site-specific runs. For example, the upper bound WCR for the peak concentration at
28 the Foresthill site is 7.36×10^{-4} mg/L per lb a.i./acre. For the generic runs of the stream,
29 the upper bound is higher by a factor of about 11—i.e., 7.9×10^{-3} mg/L. This pattern is to
30 be expected. The generic runs are intended to encompass a wide range of weather and
31 site conditions, and the upper bounds from the generic runs are intended to be plausible
32 worse-case estimates. Notably, each of the individual generic runs (i.e., for specific
33 weather patterns and soil types) are not uniformly higher than the corresponding values
34 from the site-specific runs at Chico and Foresthill. This again follows from the nature of
35 the generic runs—i.e., nine locations representing a wide range of climates each run with
36 clay, loam, and sand soil textures. Thus, several of the sites used in the generic modeling
37 have conditions that result in less offsite transfer from the treated site than is apparent for
38 conditions at the Chico and Foresthill sites. The conditions at several of the sites in the
39 generic runs result in WCR values. As noted in the documentation for WorksheetMaker
40 (SERA 2009a), the proper development of site-specific WCR values using Gleams-
41 Driver is likely to provide more reliable estimates than the use of generic WCR values
42 given in most Forest Service risk assessments.

3.2.3.4.5. Monitoring Data

Very few monitoring data are available for lambda-cyhalothrin. Lambda-cyhalothrin is not one of the pesticides included in the NAWQA monitoring program (<http://water.usgs.gov/nawqa/pnsp/pubs/circ1291/appendix1/appendix1a.pdf>).

Weston and coworkers (Weston et al. 2004; Weston et al. 2008) monitored numerous pesticides, including lambda-cyhalothrin, in streams and ponds in California. In the study by Weston et al. (2004), pesticide concentrations were assayed in areas of California with heavy agricultural uses of pesticides. The highest assayed concentration of lambda-cyhalothrin was 16.8 ng/g (16.8 µg/kg or 0.0168 mg/kg) in a tailwater pond. Lambda-cyhalothrin was detected in 12% of agricultural drainage canals, with a maximum concentration of 7.8 ng/g (7.8 µg/kg or 0.0078 mg/kg). In the study by Weston et al. (2008), pesticides were assayed in sediment from Del Puerto Creek, San Jose County, California. As in the earlier study, Del Puerto Creek is in an agricultural area in which numerous pesticides are used extensively. Lambda-cyhalothrin was assayed in sediments from this creek at concentrations of 13.1 ng/g (13.1 µg/kg or 0.0131 mg/kg) in December and 4.0 ng/g (4 µg/kg or 0.004 mg/kg) in January. Concentrations of lambda-cyhalothrin in water, however, are not reported in either of the Weston publications.

The K_d values used in the Gleams-Driver modeling (Table 5) along with the sediment concentrations (C_{Sed}) can be used to estimate the concentration of lambda-cyhalothrin in water (C_w). By definition,

$$K_d = C_{Sed} \div C_w.$$

Based on concentrations in sediment and the K_d values, the estimated concentrations in water are estimated by rearrangement of the above equation:

$$C_w = C_{Sed} \div K_d.$$

These calculations are summarized in Table 7, and the estimated average concentration in water is 4.82×10^{-6} mg/L with a range from 1.03×10^{-6} to 1.34×10^{-5} mg/L.

The estimated water concentrations from the Weston studies are not directly comparable to the water concentrations modeled using Gleams-Driver because the monitoring from Weston cannot be related to a specific application of lambda-cyhalothrin. Nonetheless, the Weston studies do reflect monitored concentrations of lambda-cyhalothrin as a result of normal agricultural use.

The average water concentration calculated from the Weston studies (4.82×10^{-6} mg/L) is virtually identical to the central estimate for Comanche Creek—i.e., 5.35×10^{-6} mg/kg (Table 6). The upper bound of 1.34×10^{-5} mg/L from the Weston studies is about a factor of 4 below the central estimate of 5.89×10^{-5} for McBride creek (Table 6). These similarities may be coincidental; nevertheless, they do suggest that concentrations in the streams at the Chico and Foresthill sites, modeled using Gleams-Driver, are of the same

1 order of magnitude as water concentrations estimated from monitored concentrations of
2 lambda-cyhalothrin in the sediment of California surface waters.

3 **3.2.3.4.6. Downstream Contamination**

4 A final consideration in interpreting the modeling output involves the impact of
5 downstream contamination. As discussed further in the risk characterization for aquatic
6 species (Section 4.4.3), HQs for some groups of aquatic organisms are substantially
7 above the level of concern, which suggests that downstream contamination by lambda-
8 cyhalothrin could be a concern. Down-stream dilution is a common problem and some
9 data are available for lambda-cyhalothrin (Bennett et al. 2005; Moore et al. 2001).

10
11 Bennett et al. (2005) studied the downstream dilution of lambda-cyhalothrin in
12 agricultural drainage ditches and noted that concentrations decreased from initial levels
13 of 347 µg/L at the point of runoff to 5.23 µg/L at 200 meters downstream—i.e., a
14 decrease in concentration by a factor of about 66. A somewhat greater decrease—i.e., a
15 factor of about 90—was noted for bifenthrin, another pyrethroid. The decrease in
16 concentration appeared to follow a log-linear (reduction-distance) relationship. At 1000
17 meters downstream, the concentration of lambda-cyhalothrin could be reduced by a factor
18 of about 100,000 (Bennett et al 2005, Figure 3, p. 2125). As discussed by Bennett et al.
19 (2005), the decreases in concentration were associated primarily with binding to and
20 degradation by aquatic macrophytes.

21
22 In the study by Moore et al. (2001), lambda-cyhalothrin was monitored in a heavily
23 vegetated agricultural drainage ditch following the release of lambda-cyhalothrin as a
24 simulated runoff event. Downstream concentrations of lambda-cyhalothrin were reduced
25 by a factor of about 10 at 45 meters downstream and to below the limit of detection at 50
26 meters downstream of the simulated release (Moore et al. 2001, Table 2). As in the study
27 by Bennett et al. (2005), however, the substantial reductions in lambda-cyhalothrin
28 concentrations in water were associated with uptake and degradation of lambda-
29 cyhalothrin by dense populations (i.e., 505.5 to 666.5 g/m²) of aquatic macrophytes.

30
31 Because the Chico and Foresthill sites are ephemeral streams, dense macrophyte
32 populations, such as those characterized by Bennett et al. (2005) and Moore et al. (2001),
33 are not likely to occur; accordingly, these studies are not directly applicable to the two
34 stream sites under consideration.

35
36 Nonetheless, sediment binding will contribute to a reduction of lambda-cyhalothrin in the
37 water column as contaminated water flows downstream. While the streams at the Chico
38 and Foresthill sites may not have a robust macrophyte community, sediments at the sites
39 and downstream from the sites are likely to bind lambda-cyhalothrin and contribute to a
40 reduction in the concentration of this pesticide in the water column. As discussed in
41 Section 3.2.3.4.3.1, Gleams-Driver was modified to account for sediment binding, and
42 the assumption was made that the functional depth for sediment binding would be 0.01 of
43 the water column depth. Based on the comparison of the modeled concentrations to the
44 limited available monitoring data (Section 3.2.3.4.6), the factor of 0.01 seems plausible
45 for the functional relative depth of sediment binding.

1
2 The only other factors that could be used to estimate a downstream reduction of lambda-
3 cyhalothrin in stream water are the distance from the treatment site to a downstream area
4 and the actual flow rate at or downstream of the treatment site. As summarized in
5 Table 4, the actual maximum flow volume of Comanche Creek is about 150 cubic feet
6 per second or 350,000,000 L/day. As illustrated in Figure 5, the maximum modeled flow
7 rate for Comanche Creek is about 120,000,000 L/day. This discrepancy is due to the
8 input of flow from a dam upstream of the treatment site on Comanche Creek. Because
9 maximum concentrations of lambda-cyhalothrin are modeled to occur at periods of peak
10 flow (Figure 5), it is reasonable to suggest that the actual concentrations of lambda-
11 cyhalothrin in Comanche Creek may be overestimated by the Gleams-Driver modeling
12 by a factor of about 3—i.e., $350,000,000 \text{ L/day} \div 120,000,000 \text{ L/day} \approx 2.91$. This minor
13 discrepancy is not relevant to the human health risk assessment because the modeled
14 concentrations in water are below the level of concern (Section 3.4). The discrepancy is
15 relevant to the risk characterization for aquatic species, discussed further in Section 4.4.3.

16
17 No information is available on flow volume or velocity in McBride Creek. In the
18 absence of this information, an elaboration of risks at the Foresthill site is not developed.

19
20 More generally, GLEAMS and consequently Gleams-Driver are field scale models that
21 are not designed for modeling large watersheds (SERA 2007b). Nonetheless, relatively
22 simple and conservative assumptions may be used to approximate downstream
23 contamination. For example, Gleams-Driver uses a flow velocity of 6,900 meters/day as
24 representative of a small stream (SERA 2007b). This is equivalent to about 4.2 miles [1
25 statue mile = 1609 meters]. Thus, over the course of a single day, a pesticide entering a
26 stream could travel about 4 miles downstream. The concentration of the pesticide in the
27 water, however, would diminish due to dilution by runoff water from untreated areas
28 along the stream bank. Assuming that the field area modeled by Gleams-Driver is
29 representative of downstream untreated areas, the impact of dilution can be
30 approximated.

31
32 For example, the length of the Foresthill site paralleling McBride Creek is about 1.3
33 miles. If the flow rate for McBride Creek were approximately 4.2 miles per day, the
34 dilution factor at 4.2 miles downstream would be about 3 [$4.2 \text{ miles per day} \div 1.3 \text{ miles} \approx$
35 3.23]. These types of calculations can be extended over longer periods of time and
36 greater distances from the treated site by considering the degradation of the pesticide in
37 both water and sediment. Again, however, these considerations cannot be applied to
38 either the Chico or Foresthill sites because of the lack of information on flow volumes
39 and flow rates. While downstream modeling of the watersheds at both sites could be
40 made in a refined risk assessment, this is beyond the scope of the current abbreviated risk
41 assessment. In addition, without at least some information or reasonably estimates of
42 actual flow rates and flow velocities, any elaborated watershed modeling could not be
43 validated.

1 **3.2.3.4.7. Concentrations in Water Used for Risk Assessment**

2 The concentrations used in this risk assessment are based on the modeling of the creeks at
3 the Chico and Foresthill sites. The expected concentrations are summarized in the upper
4 portion of Table 6. As specified in Table 6, the application rates and schedules on which
5 the modeling estimates are based are those anticipated in the Forest Service programs at
6 the two sites—i.e., 0.08 lb a.i./acre, six applications at 2-week intervals starting in May or
7 0.16 lb a.i./acre, three applications at 4-week intervals starting in May.

8
9 For more general applications, the water contamination rates (WCRs) from the generic
10 Gleams-Driver modeling are included in the database for the Forest Service’s Worksheet
11 Maker program. These values are given in the last entry in the bottom section of Table 6.
12 Differences between the generic estimates and the site-specific estimates are discussed in
13 Section 3.2.3.4.3.3.

14 **3.2.3.5. Oral Exposure from Contaminated Fish**

15 This risk assessment includes three sets of exposure scenarios for the consumption of
16 contaminated fish, and each set includes separate estimates for the general population and
17 subsistence populations. These exposure scenarios consist of one set for acute exposures
18 following an accidental spill (Worksheets D08a and D08b), another set for acute
19 exposures based on expected peak concentrations (Worksheets D08c and D08d), and the
20 third set for chronic exposures based on estimates of longer-term concentrations in water
21 (Worksheets D09a and D09b). The two worksheets in each of these three sets are
22 intended to account for different rates of wild-caught fish consumption in both general
23 and subsistence populations. Details of exposure scenarios involving the consumption of
24 contaminated fish are provided in Section 3.2.3.5 of SERA (2007a).

25
26 The consumption of contaminated fish is a standard exposure scenario used in most
27 Forest Service risk assessments. The scenario is maintained in the current risk
28 assessment in order to make the assessment more generically useful. For the two sites
29 that are the focus of the current Forest Service risk assessment, the ephemeral streams are
30 not likely to support populations of fish that humans would consume.

31 **3.2.3.6. Dermal Exposure from Swimming in Contaminated Water**

32 Conceptually and computationally, the exposure scenario for swimming in contaminated
33 water is virtually identical to the contaminated gloves scenario used for workers (Section
34 3.2.2.2)—i.e., a portion of the body is immersed in an aqueous solution of the compound
35 at a fixed concentration for a fixed period of time. The major differences in the two
36 scenarios involve the pesticide concentration in water and the exposed surface area of the
37 body. For the worker wearing contaminated gloves, the assumption is made that both
38 hands are exposed to the field solution—i.e., the concentration of the compound in the
39 applied solution. For the swimmer, the assumption is made that the entire surface area of
40 the body is exposed to the expected peak concentrations in ambient water (Table 6).
41 Also, like the exposure scenario involving contaminated gloves, the swimming scenario
42 is conservative in that it assumes zero-order absorption directly from the water to the
43 systemic circulation. Although the swimmer will not be immersed for 1 hour, the entire

1 body surface is used both as a conservative approximation (i.e., the MEI) and to consider
2 intermittent episodes during which the whole body might be immersed or at least wet.

3
4 As in the corresponding worker exposure scenario, the 1-hour period of exposure is
5 somewhat, but not completely, arbitrary, given that longer periods of exposure are
6 plausible. Nonetheless, the 1-hour period is intended as a unit exposure estimate. In
7 other words, the exposure and consequently the risk will increase linearly with the
8 duration of exposure, as indicated in Worksheet D11. Thus, a 2-hour exposure would
9 lead to a hazard quotient that is twice as high as that associated with an exposure period
10 of 1 hour. In cases in which this or other similar exposures approach a level of concern,
11 further consideration is given to the duration of exposure in the risk characterization
12 (Section 3.4). For lambda-cyhalothrin, the levels of exposure are well below the level of
13 concern.

14
15 As with the exposure scenario for the consumption of fish, the swimming scenario is
16 standard in most Forest Service risk assessments and is maintained in the current risk
17 assessment only to increase the general utility of this document to the Forest Service. For
18 the two specific sites that are the focus of this analysis, it does not seem plausible that the
19 streams in question would be used for swimming.

20 ***3.2.3.6. Oral Exposure from Contaminated Vegetation***

21 Forest Service risk assessments typically include standard exposure scenarios for the
22 acute and longer-term consumption of contaminated vegetation. Two sets of exposure
23 scenarios are provided: one for the consumption of contaminated fruit and the other for
24 the consumption of contaminated vegetation. These scenarios are detailed in Worksheets
25 D03a and D03b for acute exposure and Worksheets D04a and D04b for chronic exposure.
26 The computational details for this exposure scenario are discussed in SERA (2007a,
27 Section 3.2.3.6).

28
29 Typically, these exposure scenarios assume that the fruit and vegetation are directly
30 sprayed at the nominal application rate, and this assumption is maintained in the EXCEL
31 workbooks for lambda-cyhalothrin which are generated by WorksheetMaker. For the
32 Foresthill site, the assumption is not reasonable, given that neither edible vegetation nor
33 fruit exists in the treatment area. For the Chico site, blackberries grow along the banks of
34 Comanche Creek. As discussed in Section 3.2.3.4.3.1, applications of lambda-
35 cyhalothrin may occur within 50 feet of Comanche Creek. Thus, for these exposure
36 scenarios at the Chico site, the upper bound estimate of drift used in the Gleams-Driver
37 modeling—i.e., 0.02—is used rather than the assumption of direct spray in Worksheets
38 D03a through D04b.

39
40 For the Foresthill site, no information is available on the occurrence of edible fruit or
41 vegetation in the vicinity of the treatment area. Consequently, no modifications are made
42 to the EXCEL workbook for the Foresthill site, and the exposure scenario involving
43 direct spray of fruit and edible vegetation is used. Nonetheless, because the Foresthill
44 site is in a remote location, these exposure scenarios may be unlikely to occur, as
45 discussed further in the risk characterization for human health (Section 3.4).

1
2 The concentration of the pesticide on contaminated fruit and vegetation is estimated using
3 the empirical relationships between application rate and concentration on different types
4 of vegetation (Fletcher et al. 1994). The rates given by Fletcher et al. (1994) are based on
5 a reanalysis of data originally compiled by Hoerger and Kenaga (1972) and represent
6 estimates of the concentration in different types of vegetation (mg chemical/kg
7 vegetation) after a normalized application rate of 1 lb a.i./acre. The residue rates
8 recommended by Fletcher et al. (1994) are given in Table 8 of the current Forest Service
9 risk assessment. Note that Fletcher et al. (1994) as well as Hoerger and Kenaga (1972)
10 give only central estimates and upper bound estimates of residue rates. In Table 8, lower
11 bound estimates are given under the assumption that the ratio of the central estimate to
12 the upper bound estimate will be identical to the ratio of the lower bound to the central
13 estimate (i.e., the variability will be log-symmetrical).

1 3.3. DOSE-RESPONSE ASSESSMENT

2 3.3.1. Overview

3 The toxicity values used to develop the dose-response assessments in the human health
4 risk assessment of lambda-cyhalothrin are summarized in Table 9. The acute and chronic
5 RfDs for lambda-cyhalothrin are adopted directly from the most recent risk assessment
6 by U.S. EPA (U.S. EPA/OPP 2003b). Both the acute and chronic RfDs are based on a
7 chronic study in dogs in which the acute NOAEL was 0.5 mg/kg bw/day and the chronic
8 NOAEL was 0.1 mg/kg bw/day. Both NOAELs are based on signs of neurotoxicity,
9 transient signs of the neurotoxicity observed on Day 2 at 3.5 mg/kg bw/day and longer-
10 term signs of neurotoxicity from Week 2 to 9 observed at 0.5 mg/kg bw/day. Both RfDs
11 are derived using an uncertainty factor of 100.

12 3.3.2. Acute RfD

13 U.S. EPA/OPP sometimes derives an acute RfD for pesticide exposures that occur in a
14 single day. Accordingly, acute RfDs are usually based on developmental studies in
15 which an adverse effect is associated with a single dose of a pesticide. For lambda-
16 cyhalothrin, however, the U.S. EPA/OPP uses a somewhat different and more
17 conservative approach. As summarized in Section 3.1.9.1, the lowest developmental
18 NOAEL is 15 mg/kg bw/day, from a standard teratology study in rats. In the chronic
19 toxicity study in dogs, however, ataxia was noted in dogs from Day 2 onward at doses of
20 3.5 mg/kg bw/day. The NOAEL for ataxia was 0.5 mg/kg bw/day. Because this effect
21 was noted so early in the chronic dogs study, the dose of 0.5 mg/kg bw/day was classified
22 as an acute NOAEL, which the EPA uses as the basis for the acute RfD (U.S. EPA/OPP
23 2003b). While this approach is somewhat atypical, the dog study NOAEL of 0.5 mg/kg
24 bw/day appears to reflect the most sensitive endpoint (neurotoxicity) as well as the most
25 sensitive species (dogs).

26
27 In some Forest Service risk assessments on weak acids, dogs are considered to be
28 atypically sensitive because dogs have a very limited ability to excrete weak acids,
29 relative to other mammals, including humans. Thus, NOAELs from studies in dogs are
30 not used as the basis for RfDs. Lambda-cyhalothrin, however, is not a weak acid, and the
31 EPA decision to use the dog study as the basis for the acute RfD (U.S. EPA/OPP (2003b)
32 seems appropriate.

33
34 The acute RfD of 0.005 mg/kg bw/day is derived by dividing the acute NOAEL of 0.5
35 mg/kg bw/day by a factor of 100, the product of an uncertainty factor of 10 for
36 uncertainties in species-to-species extrapolation and an uncertainty factor of 10 for
37 uncertainties in considering sensitive individuals. These are standard uncertainty factors
38 used in most risk assessments (SERA 2007a). The acute RfD derived by U.S. EPA/OPP
39 (2003b) is used in the current Forest Service risk assessment to characterize the potential
40 risks associated with all acute exposures in workers and members of the general public.

1 **3.3.3. Chronic RfD**

2 As discussed in the previous subsection, dogs appear to be more sensitive than other
3 groups of mammals to lambda-cyhalothrin; accordingly, U.S. EPA/OPP (2003b) bases
4 both the acute and chronic RfD on the chronic dog study, in which, signs of neurotoxicity
5 were noted at 0.5 mg/kg bw/day early during the chronic exposure (i.e., from Week 2
6 through Week 9 of the 1-year study). These effects did not persist through the 1-year
7 period of exposure, and the EPA elected to classify the 0.5 mg/kg bw/day dose as a
8 LOAEL for chronic toxicity. The chronic NOAEL was set at the lowest dose tested (i.e.,
9 0.1 mg/kg bw/day). The RfD of 0.001 mg/kg bw/day was derived by dividing this
10 chronic NOAEL by an uncertainty factor of 100. The rationale for this uncertainty factor
11 is identical to that used for the acute RfD.

12
13 Following normal practice, the current Forest Service risk assessment adopts the EPA
14 chronic RfD of 0.001 mg/kg bw/day, and uses it to characterize risks for workers and
15 members of the general public associated with all longer-term exposure scenarios.

16 **3.3.4. Surrogate RfD for Occupational Exposures**

17 The U.S. EPA will sometimes derive RfDs for occupational exposures that differ from
18 the chronic RfD. This is not the case for lambda-cyhalothrin. Consequently, the chronic
19 RfD is used for general exposure scenarios in workers (i.e., non-accidental exposures
20 associated with the routine application of lambda-cyhalothrin). As discussed further in
21 Section 3.4.2 (risk characterization for workers), the use of the chronic RfD for general
22 exposures in workers does not impact the risk assessment for lambda-cyhalothrin because
23 all general exposures in workers are below the level of concern.

24 **3.3.5. Dose-Severity Relationships**

25 Unless all hazard quotients are below the level of concern (HQ=1), Forest Service risk
26 assessments attempt to define dose-severity relationships in order to more fully interpret
27 the plausible consequences of exceeding the RfD. The current abbreviated risk
28 assessment, however, does not involve the same level of review used in most Forest
29 Service risk assessments. Consequently, no formal or detailed dose-severity assessment
30 is considered.

31
32 For the assessment of the Chico and Foresthill sites, this limitation is not significant due
33 to the nature of the risk characterization and plausibility of exposures, as discussed
34 further in the risk characterization (Section 3.4).

1 **3.4. RISK CHARACTERIZATION**

2 **3.4.1. Overview**

3 Under the application conditions being considered by the Forest Service, there is no basis
4 for asserting that workers are likely to be at risk in the normal application of lambda-
5 cyhalothrin at the Chico or Foresthill sites. The risks associated with accidental
6 exposures are more difficult to characterize because of uncertainties associated with
7 estimates of some dermal exposure rates. Lambda-cyhalothrin may cause numbness or
8 tingling of the skin. If this effect is noted in workers involved in the application of
9 lambda-cyhalothrin, this would suggest that the workers are at risk of systemic toxic
10 effects. If workers effectively implement prudent worker protection measures, however,
11 the risks of systemic toxicity are probably low.

12
13 For members of the general public, the quantitative risk characterizations differ between
14 the Chico and Foresthill sites. These differences, however, may simply reflect the fact
15 that the Chico site is much better characterized than is the Foresthill site, at least in terms
16 of the plausible exposures for members of the general public. At the Chico site, the most
17 plausible exposure scenario involves the consumption of contaminated blackberries from
18 bushes that grow along the banks of Comanche Creek. The HQs for these exposure
19 scenarios are below the level of concern by a factor of at least 10. The only other non-
20 accidental exposure scenario that may be plausible involves the consumption of fish by
21 subsistence populations. The upper bound HQ for this scenario is below the level of
22 concern by a factor of 3. Much higher HQs are derived for the Foresthill site because
23 very conservative default exposure assumptions are used in the absence of specific
24 information that would justify the use of more realistic exposure assumptions. For the
25 Foresthill hill site, the HQs for the consumption of contaminated vegetation, fruit, and
26 fish exceed the level of concern by factors of 4-25. Because the Foresthill site is in a
27 relatively remote location, the risk characterization for this site may reflect potential
28 rather than plausible risk. Accidental exposure scenarios for both sites result in HQs that
29 substantially exceed the level of concern. This is typical of risk characterizations for
30 many pesticides covered by Forest Service risk assessments. In the event of major
31 accidental spills or other accidental events, remedial actions to reduce and limit
32 exposures to members of the general public would be appropriate.

33 **3.4.2. Workers**

34 For applications at the Chico and Foresthill sites, none of the HQs for general exposures
35 to workers exceeds the level of concern (HQ=1). As described in Section 3.2.2.1, the
36 term general exposure refers to the range of exposure levels to be expected during the
37 normal application of the pesticide. At the Chico site, the upper bound of the HQ is 0.1,
38 below the level of concern by a factor of 10. As discussed in Section 3.2.2.1.1, the
39 methods used to calculate worker exposure are based on standard methods used in Forest
40 Service risk assessments to estimate absorbed dose with a modification to consider the
41 use of closed cabins in airblast applications. Using a very different method—i.e., the
42 Pesticide Handlers Exposure Database developed by the U.S. EPA—very similar
43 estimates of worker exposures can be made. Thus, confidence in the risk characterization

1 is reasonably high. At the Foresthill site, the upper bound of the HQ is 0.6, below the
2 level of concern by a factor of about 2. As with workers at the Chico site, the exposure
3 assessment for workers at the Foresthill site is based on standard Forest Service estimates
4 of absorbed dose modified to account for the use of personal protective equipment
5 (Section 3.2.2.1.2). These exposure rates are supported by a study of absorbed doses in
6 workers involved in ground applications of lambda-cyhalothrin. Consequently,
7 confidence in the risk characterization of workers at the Foresthill site is also high. There
8 is no basis for asserting that workers are likely to be at risk in the normal application of
9 lambda-cyhalothrin under the exposure conditions at the Chico or Foresthill sites. Note,
10 however, that this risk characterization is based on the assumptions that the worker
11 protection measures described by the Forest Service are rigorously and effectively
12 implemented.

13
14 Confidence in the risk characterization for accidental exposure scenarios is mixed. As
15 indicated in Worksheet E02 in the EXCEL workbooks that accompany this risk
16 assessment (Attachments 1 and 2), the HQs for exposure scenarios associated with
17 wearing contaminated gloves are much higher (a maximum HQ of 46) than those for
18 exposure scenarios involving accidental spills onto the hands or lower legs (a maximum
19 HQ of 0.004). The scenarios for spills are based on a human study of the dermal
20 absorption rate of lambda-cyhalothrin. Confidence in these scenarios is high. As
21 discussed in Section 3.1.3.2, however, the measured human dermal absorption rate is
22 much lower than rates based on QSAR estimates of the first-order dermal absorption rate,
23 which calls into question the estimates of K_p values used to estimate the absorbed doses
24 associated with the contaminated glove scenarios. The scenarios for contaminated gloves
25 are based on the assumption of zero-order absorption using the K_p derived from an
26 algorithm developed by U.S. EPA/ORD (1992); nevertheless, confidence in this
27 algorithm is low because the lipophilicity of lambda-cyhalothrin exceeds that of the
28 compounds used to develop the algorithm. Thus, there is little confidence in the
29 relatively high HQs associated with the contaminated glove scenarios.

30
31 Regardless of the variability in confidence in the accidental exposure scenarios, the most
32 plausible outcome of prolonged dermal exposure to lambda-cyhalothrin is numbness or
33 tingling of the skin (paresthesia). If this effect is noted in workers involved in the
34 handling or application of lambda-cyhalothrin, it is most certainly because the workers
35 are not effectively implementing worker protection practices, thereby putting themselves
36 at risk of systemic toxic effects.

37 **3.4.3. General Public**

38 At the Chico site, the probability of exposure to lambda-cyhalothrin is high because the
39 site is open to and used by members of the general public. The risks to members of the
40 general public, however, appear to be low. For non-accidental exposures, the highest
41 HQs are associated with the consumption of contaminated vegetation—i.e., an HQ of 0.7
42 for acute exposure and 0.6 for chronic exposure. These exposure scenarios are standard
43 in all Forest Service risk assessments, but the associated HQs appear to be irrelevant for
44 the Chico site, given the lack of edible vegetation at the site. Fruit, specifically
45 blackberries, which might be consumed by members of the general public, do grow at

1 this site; however, the HQs for these scenarios are remarkably low—upper bound HQ
2 values of 0.1 for acute exposure and 0.08 for longer-term exposures. The only other non-
3 accidental HQ that approaches a level of concern involves the consumption of
4 contaminated fish by subsistence populations—i.e., an upper bound value of 0.3. As
5 noted in Section 3.2.3.5, it is not likely that these HQ values are of any practical concern
6 because this ephemeral stream is not likely to be used as a source of fish.

7
8 The Foresthill site is in a remote location. Although members of the general public are
9 not specifically excluded from this site, the probability that they would access this site
10 and be exposed to significant amounts of lambda-cyhalothrin appears to be much lower
11 than the probability of such exposures at the Chico site. Nonetheless, the HQs for
12 members of the general public are much higher at the Foresthill site than at the Chico
13 site. The highest upper bound HQs are associated with the consumption of contaminated
14 vegetation (acute HQ = 35, chronic HQ = 28), contaminated fruit (acute HQ = 5, chronic
15 HQ = 4), and the acute scenarios for the consumption of contaminated fish by subsistence
16 populations (acute HQ = 4).

17
18 It is not clear that the Foresthill site has vegetation or fruit likely to be consumed by
19 members of the general public. In addition, the scenarios for the consumption of
20 vegetation and fruit at the Foresthill site all assume direct spray. This is a standard and
21 conservative assumption in Forest Service risk assessments. For the Chico site, this
22 assumption is not reasonable based on information for the Chico site. For the Foresthill
23 site, the assumption of direct spray may or may not be reasonable.

24
25 McBride Creek at the Foresthill site does not support populations of fish which might be
26 consumed by humans (Bakke 2010). As discussed in Section 3.2.3.4.3.2, information
27 from the Forest Service indicates that McBride Creek will be dry during the periods when
28 lambda-cyhalothrin is applied, and this assertion is supported by Gleams-Driver
29 modeling.

30
31 Some of the accidental exposure scenarios lead to HQs that substantially exceed the level
32 of concern—i.e., upper bound HQs of about 1500 for the consumption of fish by
33 subsistence populations, 300 for fish consumption by the general population, and 3 for
34 the consumption of contaminated water by a small child.

35
36 For the Foresthill site, none of these HQs appears to be relevant. McBride Creek is likely
37 to be dry when lambda-cyhalothrin applications are made.

38
39 For the Chico site, however, information from the Forest Service indicates that pools of
40 standing water may occur in the Comanche Creek bed during periods when lambda-
41 cyhalothrin is applied. Nonetheless, the consumption of contaminated fish does not
42 appear to be a relevant scenario for the Chico site. As discussed in Section 4.4.3.1 (risk
43 characterization for fish), an accidental spill of a large amount of lambda-cyhalothrin into
44 a small volume of water is likely to kill all fish in the water. The consumption of
45 contaminated water by a small child after an accidental spill is a more plausible
46 accidental exposure scenario. In the event of a spill of lambda-cyhalothrin into a small

1 body of water, measures should be taken to ensure that members of the general public do
2 not consume the water. This recommendation, of course, is standard for any accidental
3 spill of any pesticide into water.

4 **3.4.4. Sensitive Subgroups**

5 For exposures to almost any chemical, there is particular concern for children, women
6 who are pregnant or may become pregnant, the elderly, or individuals with any number of
7 different diseases. As discussed in Section 3.1.3, lambda-cyhalothrin is detoxified in the
8 liver and metabolites are excreted primarily by the kidney. It is possible that individuals
9 with liver or kidney diseases could be more sensitive to lambda-cyhalothrin than other
10 individuals. This statement is true for most pyrethroids (ATSDR 2003).

11 **3.4.5. Connected Actions**

12 The Council on Environmental Quality (CEQ), which provides the framework for
13 implementing NEPA, defines connected actions (40 CFR 1508.25) as actions which
14 occur in close association with the action of concern; in this case, the use of a pesticide.
15 Actions are considered to be connected if they: (i) Automatically trigger other actions
16 which may require environmental impact statements; (ii) Cannot or will not proceed
17 unless other actions are taken previously or simultaneously, and (iii) Are interdependent
18 parts of a larger action and depend on the larger action for their justification. Within the
19 context of this assessment of lambda-cyhalothrin, “connected actions” include actions or
20 the use of other chemicals which are necessary and occur in close association with use of
21 lambda-cyhalothrin.

22
23 As discussed in detail in Sections 3.1.14, lambda-cyhalothrin formulations contain inert
24 components. The inert ingredients in several lambda-cyhalothrin formulations are not
25 well characterized. The inerts that are disclosed, such as petroleum hydrocarbons, may
26 cause a wide spectrum of toxic effects. The limited data on the toxicity of the
27 formulations do not yield a consistent pattern in terms of the potential impact of the inert
28 ingredients on the toxicity of the formulations. Thus, the impact of the inerts on the
29 potential risks in using different formulations cannot be well characterized.

30 **3.4.6. Cumulative Effects**

31 Similar to the issues involved in assessing inerts, it is beyond the scope of the current risk
32 assessment to identify and consider all agents that might interact with, or cause
33 cumulative effects with lambda-cyhalothrin. To do so quantitatively would require a
34 complete set of risk assessments on each of the other agents to be considered.

35
36 U.S. EPA/OPP (2009c) groups lambda-cyhalothrin with three other pyrethroids:
37 cyfluthrin, cypermethrin, and deltamethrin (all Type II pyrethroids). Exposures to
38 mixtures of any combination of these four pyrethroids would be regarded as additive, in
39 terms of risk. Esfenvalerate is classified as having mixed (Type I/II) activity. It would
40 be prudent to regard risks associated with esfenvalerate and lambda-cyhalothrin as
41 additive.

1 The current Forest Service risk assessment does consider the effect of repeated exposures
2 to lambda-cyhalothrin for both workers and members of the general public. The chronic
3 RfD is used as an index of acceptable longer-term exposures. Consequently, the risk
4 characterizations presented in this risk assessment for longer-term exposures specifically
5 address and encompass the potential impact of the cumulative effects of repeated human
6 exposure to lambda-cyhalothrin.

1 4.1. HAZARD IDENTIFICATION

2 4.1.1. Overview

3 As with most commercial insecticides, lambda-cyhalothrin is much more toxic to insects
4 and other arthropods than it is to vertebrates. In terms of terrestrial organisms, lambda-
5 cyhalothrin is highly toxic to insects with contact LD₅₀ values as low as 0.065 mg/kg
6 bw. Non-arthropod invertebrates appear to be substantially less sensitive than terrestrial
7 insects or groups of arthropods to lambda-cyhalothrin. Also, lambda-cyhalothrin is
8 classified as only moderately toxic to mammals. The lowest reported oral LD₅₀ for
9 technical grade lambda-cyhalothrin is 56 mg/kg bw, a factor of over 800 below the
10 lowest contact LD₅₀ in insects. Birds are much less sensitive than either insects or
11 mammals to lambda-cyhalothrin, which is classified as only slightly toxic to birds.
12 Canids appear to be somewhat more sensitive than rodents to lambda-cyhalothrin.
13 Similarly, mallards appear to be somewhat more sensitive than quail to lambda-
14 cyhalothrin.

15
16 Figure 6 illustrates an overview of the acute toxicity data for aquatic organisms, including
17 fish, aquatic arthropods, and other aquatic invertebrates. The specific data used to
18 develop this plot are discussed in the subsections on fish (Section 4.1.3.1) and aquatic
19 invertebrates (Section 4.1.3.3). As illustrated in Figure 6, aquatic arthropods are more
20 sensitive than fish. Based on the geometric mean of the 96-hour LC₅₀ values in fish (0.5
21 µg/L) and 48-hour LC₅₀ values in aquatic arthropods (0.08 µg/L), arthropods are more
22 sensitive than fish by a factor of about 6. Relatively little data are available on the
23 toxicity of lambda-cyhalothrin to non-arthropod aquatic invertebrates; however, the
24 available data suggest that non-arthropod aquatic invertebrates (such as mollusks) are
25 much less sensitive than either aquatic arthropods or fish to lambda-cyhalothrin. For
26 amphibians, the only available LC₅₀ value for cyhalothrin is 4 µg/L in a tadpole. Aquatic
27 plants appear to be unaffected by lambda-cyhalothrin.

28 4.1.2. Toxicity to Terrestrial Organisms

29 4.1.2.1. Mammals

30 As discussed in the human health risk assessment (Section 3.1), there are several toxicity
31 studies regarding the exposure of experimental mammals to lambda-cyhalothrin. These
32 studies are also relevant to the assessment of potential hazards to mammalian wildlife.
33 Based on acute oral LD₅₀ values of 56-79 mg/kg bw (Section 3.1.4), U.S. EPA/EFED
34 (1998) classifies lambda-cyhalothrin as *Moderately Toxic* to mammals.

35
36 The ecological risk assessment attempts to identify subgroups of mammals that may
37 display greater or lesser sensitivity to a particular pesticide. These differences may be
38 based on allometric scaling (e.g., Boxenbaum and D'Souze 1990) or differences in
39 physiology. As discussed in Section 3.3 (dose-response assessment for human health
40 effects), dogs appear to be somewhat more sensitive than rodents lambda-cyhalothrin.
41 Data on mammals, however, are insufficient to determine if this difference in sensitivity
42 is related to body size, physiological differences, or simply to random variability among
43 different studies conducted at different times in different laboratories. For the current

1 Forest Service risk assessment, canids are assumed to be more sensitive than rodents or
2 other mammals; accordingly, different toxicity values are derived for canids and non-
3 canid species (Section 4.3.2.1). In the absence of a clear relationship between body
4 weight and toxicity across a range of mammalian species, however, separate toxicity
5 values are not derived for small and large non-canid mammals.

6 **4.1.2.2. Birds**

7 The open literature on lambda-cyhalothrin does not include avian toxicity studies or field
8 studies. Typically, the EPA requires three types of avian toxicity studies for pesticide
9 registration: single gavage dose LD₅₀ studies, 5-day dietary toxicity studies, and chronic
10 (≈30 week) dietary reproduction studies. The required studies are usually conducted with
11 mallard ducks and bobwhite quail.

12
13 For lambda-cyhalothrin, there is only a gavage LD₅₀ in which Roberts and Fairley (1984)
14 assayed the toxicity of 739, 1040, 1620, 2580, or 3950 mg/kg technical grade lambda-
15 cyhalothrin in mallards. There was no mortality at any dose, and the only potential sign
16 of toxicity was a small liver lesion, which was not attributed to treatment, in one bird in
17 the 2580 mg/kg bw dose group.

18
19 Four 5-day dietary studies were submitted to the EPA (Roberts et al. 1981a,b; Roberts et
20 al. 1985a,b). In the earlier studies by Roberts et al. (1981a,b), no clear dose-response
21 relationships were observed, and these studies were repeated by Roberts et al. (1985a,b).
22 Roberts et al. (1985a) assayed dietary concentrations of 577, 1020, 1980, 3040, 4090, or
23 5300 ppm lambda-cyhalothrin in quail. No significant mortality or other signs of toxicity
24 were noted at any dose. Mortality was noted in 1/10 birds sporadically—i.e., 10%
25 mortality at 577, 1020, 3040, and 5300 ppm, but 0% mortality was noted at 1980 and
26 4090 ppm.

27
28 Mallards appear to be somewhat more sensitive than quail to lambda-cyhalothrin. In the
29 study by Roberts et al. (1985b), mallards were exposed to lambda-cyhalothrin at dietary
30 concentrations of 0, 505, 1030, 2030, 3020, 4020, or 5040 ppm for 5 days. The dietary
31 LC₅₀ was 3948 ppm. At a concentration of 2030 ppm, 2 of 10 birds died. While this
32 incidence is not statistically significant from the control group ($p=0.2368$) using the
33 Fisher Exact test, the small number of animals tested makes standard statistical tests
34 relatively insensitive. At 505 ppm, no mortality was noted. At higher doses, signs of
35 neurotoxicity were noted—i.e., subdued behavior and unsteady gait—as well as a
36 decrease in body weight gain and food consumption. This study was classified as core by
37 U.S. EPA/OPP. Based on these acute toxicity studies, lambda-cyhalothrin is classified as
38 *slightly toxic* to birds (U.S. EPA/EFED 1988).

39
40 Three reproduction studies were submitted to the U.S. EPA, two in mallards (Beavers et
41 al. 1989; Roberts et al. 1982a) and one in quail (Roberts et al. 1982b). As in the acute
42 dietary studies, mallards appear to be somewhat more sensitive than quail to lambda-
43 cyhalothrin. The studies by Roberts et al. (1982a,b) involved only two dietary
44 concentrations of lambda-cyhalothrin, 5 and 50 ppm. In the quail study, Roberts et al.
45 (1982b) do not report adverse effects on adults or offspring. In the study on mallards,

1 however, Roberts et al. (1982a) report decreased egg production at 50 ppm. In the later
2 study on mallards, Beavers et al. (1989) used dietary concentrations of 0.5, 5, 15, and 30
3 ppm as well as a control group. The only effects clearly associated with treatment were
4 increases in food consumption in the 15 and 30 ppm groups. These increases were not
5 accompanied by statistically significant increases in body weight.

6
7 In the ecological risk assessment conducted by U.S. EPA/EFED (1988) the following
8 comment is made on the reproduction study by Roberts et al. (1982a):

9
10 *Technical PP321 is expected to be even more toxic since it is the*
11 *"biologically active constituent" of cyhalothrin. Therefore, the*
12 *NOEL is probably less than 5 ppm.*

13 U.S. EPA/EFED (1988, p. 8)

14
15 The above comment suggests that the compound tested by Roberts et al. (1982b) was
16 cyhalothrin (i.e., the mixture of four enantiomers) rather than lambda-cyhalothrin (the
17 mixture of two enantiomers). The genesis of this comment is not clear. The DER for
18 Roberts et al. (1982b) identifies the test substance as PP321. This same designation is
19 also used in the DER for the study by Beavers et al. (1989), which explicitly identifies the
20 test compound as lambda-cyhalothrin.

21 **4.1.2.3. Reptiles**

22 There are no data regarding the toxicity of lambda-cyhalothrin to reptiles or amphibians
23 in the database maintained by Pauli et al. (2000), and there are no other sources of such
24 data in the available literature.. Generally, in the absence of toxicity data concerning
25 reptile exposure to pesticides, the EPA recommends the use of birds as suitable
26 surrogates for reptiles.

27 **4.1.2.4. Terrestrial Invertebrates**

28 Appendix 4 summarizes laboratory assays (Appendix 4, Table 1) and field studies
29 (Appendix 4, Table 2) on the toxicity of lambda-cyhalothrin to terrestrial invertebrates.

30
31 There is a range of laboratory studies concerning the toxicity of lambda-cyhalothrin to
32 terrestrial invertebrates. In Appendix 4 (Table 1), these studies are classified as direct
33 contact, residue contact (i.e., contact with lambda-cyhalothrin residues on surfaces),
34 dietary, and soil bioassays. Although the EPA does not typically quantify risks to
35 terrestrial invertebrates, it does require both direct contact and oral toxicity studies on
36 honeybees, for pesticide registration. As summarized in Appendix 4, the studies
37 submitted to the EPA yield a direct contact LD₅₀ of 0.038 µg/bee and a dietary LD₅₀ of
38 0.909 µg/bee. Based on these bioassays, direct contact is much more toxic than dietary
39 exposure in honeybees. In accordance with these studies, U.S. EPA/EFED (1988)
40 classifies lambda-cyhalothrin as *highly toxic* to honeybees.

41
42 Several other direct contact bioassays on bees and other invertebrate species are available
43 in the published literature on lambda-cyhalothrin. Because of large differences in body
44 size among terrestrial invertebrates, these studies are best compared in terms of doses

1 expressed as mg/kg bw. Neither U.S EPA/EFED (1988) nor the two published honeybee
2 studies (Johnson et al. 2006 ; Pilling and Jepson 1993) specify the weight of honeybees.
3 For these studies, doses in units of mg/kg bw are presented in Appendix 2 and are
4 calculated assuming a body weight of 93 mg (i.e., 0.000093 kg) from USDA/APHIS
5 (1993). In the contact bioassays summarized in Appendix 2, LD₅₀ values for honeybees
6 range from 0.4 mg/kg bw (U.S. EPA/OPP 1988) to 1.7 mg/kg bw (Mayer et al. 1998)—
7 i.e., the values vary by a factor of 4.25. The most sensitive species is the alfalfa leafcutter
8 bee, with an LD₅₀ of 0.065 mg/kg bw, about a factor of 6 below the LD₅₀ in honeybees
9 cited in U.S. EPA/OPP (1988). Based on the study by Dinter and Poehling (1995),
10 spiders appear to be about as sensitive as honeybees to lambda-cyhalothrin. Direct
11 contact bioassays are used in Forest Service risk assessments, to quantify risks associated
12 with direct spray and spray drift. Differences in sensitivity among invertebrates after
13 direct contact are discussed further in the dose-response assessment (Section 4.3.2.3.2).
14

15 Dietary assays are also used quantitatively in Forest Service risk assessments to evaluate
16 risks to invertebrates from the consumption of lambda-cyhalothrin. Only two studies are
17 available, the oral study in honeybees by Gough et al. (1984) and the published study by
18 Abro et al. (1997) on the tobacco cutworm, a target species. These two studies are not
19 directly comparable. Gough et al. (1984) report an oral LD₅₀ of about 0.909 µg/bee in
20 bees fed sucrose contaminated with lambda-cyhalothrin. Based on a body weight
21 assumption of 93 mg for the honeybee (USDA/APHIS 1993), the oral LD₅₀ is equivalent
22 to about 9.7 mg/kg bw [0.000909 mg/bee ÷ 0.000093 kg/bee ≈ 9.774 mg/kg bw]. The
23 study by Abro et al. (1997) involved exposures of cutworms to cauliflower dipped in
24 solutions of lambda-cyhalothrin; however, since the study does not indicate the
25 concentration of lambda-cyhalothrin on the cauliflower, it is not possible to estimate the
26 doses to the cutworms in units of mg/kg bw.
27

28 Soil assays may also be used quantitatively in Forest Service risk assessments. Most soil
29 bioassays are conducted in earthworms. As summarized in Appendix 2, earthworms
30 appear to be relatively tolerant, with acute LC₅₀ values ranging from about 24 to greater
31 than 100 ppm, chronic LC₅₀ values ranging from about 3 to 10 ppm, and EC₅₀ values for
32 avoidance ranging from about 0.2 to 3 ppm (Garcia et al. 2008). As discussed by
33 Framptom et al. (2006), arthropods appear to be much more sensitive than annelids to
34 lambda-cyhalothrin. The lowest LC₅₀ reported by Framptom et al. (2006) is 0.5 ppm for
35 an isopod, *Porcellionides pruinosus*. More recently, Ruan et al. (2009) reported adverse
36 effects—i.e., a decrease in locomotion—in soil nematodes at concentrations as low as
37 0.002 ppm and a decrease in body size at concentrations of 2 ppm. Ruan et al. (2009) do
38 not provide a dose-response assessment or a tabular summary of the data that would
39 permit the calculation of EC₅₀ values for either the decreases in locomotion or body size.
40 While not directly comparable to the other soil bioassays, the results from Ruan et al.
41 (2009) suggest that soil nematodes may be more sensitive than other groups of soil
42 invertebrates to lambda-cyhalothrin.
43

44 Several residue contact assays are also available. These studies involve application of the
45 pesticide to a surface, such as glass, filter paper, or vegetation. The exposure can be
46 expressed in units of concentration (e.g., mg/L applied to a surface) or units of mass per

1 unit area. The studies expressed in units of concentration are not used directly in Forest
2 Service risk assessments. These studies are typically focused on differences among
3 pesticides and/or differences among species or age groups. For example, the study by
4 Booth et al. (2007) demonstrates that aphids (a target species) are very sensitive to
5 lambda-cyhalothrin, while lacewings (a beneficial species that preys on aphids) are
6 relatively tolerant to lambda-cyhalothrin. The exposures that are expressed in units of
7 mass per surface area can be converted to units of application rate—i.e., lb a.i./acre.
8 These studies indicate that exposures equivalent to application rates as low as 0.0044 lb
9 a.i./acre can cause adverse effects in some species of terrestrial invertebrates (Desneux et
10 al. 2004b).

11
12 As summarized in Appendix 2 (Table 2), the literature on lambda-cyhalothrin includes
13 five field studies: Gough et al. (1986), Hearn (1985), Li and Harmsen (1993), Mayer et
14 al. (1998), and Niehoff et al. (1994). In the Mayer et al. (1998) study involving residue
15 exposure, cages of leafcutter bees were placed in fields after applications of 0.001, 0.15,
16 and 0.25 lb a.i./acre lambda-cyhalothrin. Adverse effects were not apparent at the two
17 lower application rates; however, a decrease in bee populations was noted at the highest
18 application rate. In the study by Hearn (1985), honeybees were subject to direct spray at
19 application rates of 0.0075 and 0.015 lb a.i./acre with resulting mortality rates of about 50
20 and 90%, respectively. In post-spray observations, Hearn (1985) noted a decrease in the
21 number of foraging bees on the treated alfalfa over a 2-day period, suggesting that bees
22 will avoid treated vegetation. Similar observations were made with honeybees in the
23 avoidance study by Gough et al. (1986). Li and Harmsen (1993) noted a significant
24 increase in mite populations after applications of lambda-cyhalothrin at very low rates—
25 i.e., 0.001 to 0.005 lb a.i./acre. While this effect would not typically be regarded as
26 adverse, Li and Harmsen (1993) conjecture that the reason for the increase in mite
27 population could be attributable to the repellent effect of lambda-cyhalothrin. The study
28 by Nieheff et al. (1994) noted adverse effects in rove beetles and mixed effects in
29 different species of spiders, also at very low application rates—0.0022 to 0089 lb
30 a.i./acre. This study is discussed further in the risk characterization.

31
32 Resistance is an important issue in the use of many pesticides, including pyrethroids. As
33 discussed in Section 2, the Forest Service is considering the use of lambda-cyhalothrin as
34 a replacement for or auxiliary treatment along with esfenvalerate. In terms of assessing
35 the potential for insect resistance to insecticides, all pyrethrins and pyrethroids are
36 classified as Type 3A insecticides (IRAC 2009). Insect resistance to lambda-cyhalothrin
37 and esfenvalerate in field populations of insects is quantified in three studies: Ahmad and
38 Arif (2009), Atkinson et al. (1991), and Yu and Nguyen (1992). In the study by Yu and
39 Nguyen (1992) on resistance in field populations of the diamondback moth, resistance
40 factors of almost 11,000 were noted for lambda-cyhalothrin, which is substantially
41 greater than the resistance factor of about 2300 for esfenvalerate. Ahmad and Arif (2009)
42 report a more modest difference in the maximum resistance factor in field populations of
43 the spotted bollworm in Pakistan—i.e., a factor of 286 for lambda-cyhalothrin and 108
44 for esfenvalerate. Atkinson et al. (1991) also report similar resistance factors for
45 cyhalothrin (not lambda-cyhalothrin) and esfenvalerate—i.e., 40.6 for cyhalothrin and
46 29.4 for esfenvalerate—in field populations of the German cockroach in Florida. While

1 only the study by Atkinson et al. (1991) was conducted in the United States, all three
2 studies suggest greater resistance to cyhalothrin or lambda-cyhalothrin than to
3 esfenvalerate. Hansen (2003) report that pollen beetles in Denmark appear to be resistant
4 to both lambda-cyhalothrin and esfenvalerate; however, differences in the magnitude of
5 resistance to the two pesticides cannot be determined.

6 **4.1.2.5. Terrestrial Plants (Macrophytes)**

7 Apparently, there are no studies concerning the toxicity of lambda-cyhalothrin to
8 terrestrial plants, and terrestrial plants are not addressed in the ecological risk
9 assessments conducted by U.S. EPA/OPP on lambda-cyhalothrin (U.S. EPA/EFED
10 1987c,d; 1988, 1994a,b). While the lack of information does not lead to a presumption of
11 safety, lambda-cyhalothrin has been used for many years to control insect pests on
12 vegetation. It seems reasonable to assume that adverse effects on plants would have been
13 recognized and reported if lambda-cyhalothrin were highly phytotoxic.

14 **4.1.2.6. Terrestrial Microorganisms**

15 There are two studies concerning the toxicity of lambda-cyhalothrin to terrestrial
16 microorganisms: Cycon et al. (2006) and Latif et al. (2008). Cycon et al. (2006) assayed
17 microbial activity in loam soils at lambda-cyhalothrin concentrations of 0.2, 1, or 20 ppm
18 over a 28-day period. Lambda-cyhalothrin increased the numbers of soil bacteria up to
19 Day 28 and fungi up to Day 14. At 20 ppm, lambda-cyhalothrin increased soil
20 respiration and decreased the number of denitrifiers up to Day 14. The effect on
21 denitrifiers was not significant by Day 28. Latif et al. (2008) also report an increase the
22 in the population of soil microorganisms after exposure to lambda-cyhalothrin in soil
23 over a 32-day period. The concentrations of lambda-cyhalothrin used in this study,
24 however, are not clear.

25 **4.1.3. Aquatic Organisms**

26 **4.1.3.1. Fish**

27 **4.1.3.1.1. Acute Toxicity**

28 Studies on the toxicity of lambda-cyhalothrin to fish are summarized in Appendix 3.
29 Based on 96-hour LC₅₀ values of 0.21 µg/L in bluegill sunfish and 0.24 µg/L in rainbow
30 trout, U.S. EPA/EFED (1988) classifies lambda-cyhalothrin as *very highly toxic* to fish.
31 This is the highest toxicity category used by the EPA for aquatic organisms, and it is used
32 in designating chemicals with acute LC₅₀ values of less than 0.1 mg/L or 100 µg/L (U.S.
33 EPA/EFED 2001).

34
35 Table 10 summarizes the acute toxicity values (LC₅₀ values) in fish cited by U.S.
36 EPA/EFED (1988) along with additional LC₅₀ values from the open literature. These
37 values are ranked in order of 96-hour LC₅₀ values from Appendix 3 (Table 1). Based on
38 the available data, the most sensitive species is the golden orfe, with an LC₅₀ of 0.078
39 µg/L; the most tolerant species is channel catfish with an LC₅₀ of 7.92 µg/L. Thus, the
40 range of reported LC₅₀ values varies by a factor of about 100 [7.92 µg/L ÷ 0.078 µg/L ≈
41 102]

1
2 A plot of all of the 96-hour LC₅₀ values in Table 10 is illustrated in Figure 6, along with
3 toxicity data on aquatic invertebrates. In Figure 6, the x-axis is the LC₅₀ value and the y-
4 axis is the cumulative frequency of the LC₅₀ value for each group of organisms. For
5 example, there are a total of 14 LC₅₀ values for fish. As noted above, the lowest value is
6 an LC₅₀ of 0.078 µg/L. The frequency is taken as 1-0.5/14 or 0.037. The second lowest
7 LC₅₀ value is 0.106 µg/L and this is assigned a frequency of 2-0.5/14 or 0.107. More
8 generally, the ith toxicity value has a frequency of i-0.5/N, where N is the number of
9 values in the set. Note that the x-axis is given on a logarithmic scale under the
10 assumption that generally LC₅₀ values in different groups of organisms will be log-
11 normally distributed.

12
13 A plot such as Figure 6 is sometimes referred to as a *species sensitivity distribution*
14 (SSD). Species sensitivity distributions can be used quantitatively in risk assessments
15 (e.g., Posthuma et al. 2002) as tools in probabilistic risk assessment. This technique is
16 not currently used quantitatively in Forest Service risk assessments. Nonetheless, species
17 sensitivity plots such as those presented in Figure 6 are useful for illustrating differences
18 in sensitivity among different groups of organisms.

19
20 In terms of practical significance to the current risk assessment, the study by Maund et al.
21 (1998) is particularly important. Most acute toxicity studies in aquatic organisms are
22 conducted using either pure natural water or reconstituted water. These test systems will
23 have little if any organic carbon in the water. For many water soluble pesticides (e.g.,
24 most salts of weak-acid herbicides), this distinction makes very little difference in the
25 toxicity of a compound. For highly lipophilic compounds such as lambda-cyhalothrin,
26 however, binding to organic carbon either in water or sediment can have a substantial
27 influence on the toxicity of the pesticide. Maund et al. (1998) conducted a separate
28 bioassay in carp in which lambda-cyhalothrin was applied to either the water phase (i.e.,
29 a standard bioassay) or to sediment prior to the addition of water. The 96-hour LC₅₀
30 value for carp was 4 µg/L in standard bioassay and 74 µg/L in the sediment assay. Thus,
31 in terms of the nominal exposure concentration, sediment reduced the toxicity of lambda-
32 cyhalothrin by a factor of about 20 [74 µg/L ÷ 4 µg/L ≈ 18.5]. As discussed further in
33 Section 4.1.3.3, the importance of the binding of lambda-cyhalothrin to sediment or
34 dissolved organic carbon is demonstrated in greater detail in studies on aquatic
35 invertebrates.

36
37 As noted in Section 2, lambda-cyhalothrin is a combination of two enantiomers (i.e.,
38 mirror image isomers). Xu et al. (2008) recently demonstrated that the -enantiomer is
39 much more toxic than the +enantiomer to zebrafish (*Danio rerio*). While not clearly
40 stated by Xu et al. (2008), the more toxic -enantiomer appears to correspond to gamma-
41 cyhalothrin. On the other hand, Wang et al. (2007) report very little difference in the
42 toxicity of gamma-cyhalothrin and lambda-cyhalothrin to the zebrafish. The reason for
43 these discrepancies is not apparent. Nonetheless, the toxicity values assayed in the
44 studies by Xu et al. (2008) and Wang et al. (2007) are above the most sensitive LC₅₀
45 available for lambda-cyhalothrin—i.e., 0.078 µg/L for the golden orfe, as discussed
46 above. Other studies specifically on lambda-cyhalothrin are focused on *in vitro*

1 biochemical changes in fish (e.g., Babiacuten and Tarazona 2005; Dogan 2006). These
2 studies were also conducted at concentrations substantially higher than the lowest
3 reported LC₅₀ values (i.e., >2 mg/L).

4 **4.1.3.1.2. Chronic Toxicity**

5 The two longer-term toxicity studies in fish include an early life-stage study in
6 sheepshead minnow (Hill et al. 1985d) and a full life-cycle study in fathead minnows
7 (Tapp et al. 1990). Both studies were submitted to U.S. EPA/OPP in support of the
8 registration of lambda-cyhalothrin, and data evaluation records (DERs) are available for
9 both studies. As summarized in Appendix 3 (Table 2), the early life-stage study in
10 sheepshead minnow is relatively straightforward. The study is classified as Core by U.S.
11 EPA/OPP (1988) with an NOEC of 0.25 µg/L and a LOEC of 0.38 µg/L, based on a
12 decrease in body weight of the fry.

13
14 The full life-cycle study by Tapp et al. (1990) indicates much lower toxicity values. The
15 DER for this study reports an NOEC of 0.031 µg/L and an LOEC of 0.062 µg/L. The
16 LOEC is based on an increase in body weight in male fish. While increased body weight
17 is not typically viewed as an adverse effect, comments from U.S. EPA/EFED in the DER
18 indicate that the increase in body weight at 0.062 µg/L was considered a hormetic
19 response—i.e., an interference with normal growth. No decrease in weight, however,
20 was noted at higher concentrations. Notwithstanding the arguable nature of 0.062 µg/L
21 as a LOEC based on growth, a decrease in the total number of eggs produced (about 60%
22 relative to the untreated control group) was also noted at the 0.062 µg/L concentration.
23 While this effect was not considered statistically significant, it was viewed by the U.S.
24 EPA/EFED as biologically significant. Based on the data presented in the DER (Tapp et
25 al. 1990, p. 18), the classification of 0.062 µg/L as an LOEC appears reasonable.
26 Subsequent summaries from EFED repeat the classifications of 0.031 µg/L as the chronic
27 NOEC and 0.062 µg/L as the chronic LOEC (U.S. EPA/EFED 1994b).

28 **4.1.3.1.3. Field and Mesocosm Studies**

29 In addition to the acute and chronic studies on fish, Appendix 3 (Table 3) also
30 summarizes three field/microcosm studies. Two studies (Hamer et al. 1994; Lawler et al.
31 2003) involved applications of lambda-cyhalothrin to rice paddies. Hamer et al. (1994)
32 summarizes the results of a field study conducted in the Philippines in which no adverse
33 effects were noted in fish at lambda-cyhalothrin concentrations of about 12.5 µg/L. In a
34 study conducted in California, Lawler et al. (2003) do not provide monitoring data on
35 concentrations of lambda-cyhalothrin in water. They do, however, report an application
36 rate of 5.8 g/ha, equivalent to 0.58 mg/m². While the average depth of the water in the
37 rice paddies is not specified, Lawler et al. (1994) make the following comment on
38 another study: *water depth was as low as 10 cm, about half the depth of our study.*
39 Assuming a water depth of about 20 cm (0.2 meters), the nominal concentration in the
40 rice paddy immediately after treatment would be about 2.9 µg/L. Unlike the study by
41 Hamer et al. (1994), Lawler et al. (2003) note substantial fish mortality and cite their
42 results as ... *the 1st to document fish kills from lambda-cyhalothrin applied at label rates*
43 *under field conditions* (Lawler et al. 2003, p. 431). The reason for the discrepancies
44 between these two studies is not apparent. As summarized in Table 10 and discussed

1 above, acute LC₅₀ values for lambda-cyhalothrin in fish range from 0.078 to 7.92 µg/L.
2 These LC₅₀ values are consistent with the report of fish mortality in the study by Lawler
3 et al. (2003).

4
5 The study by Hill et al. (1994) involved mesocosms with peak concentrations of lambda-
6 cyhalothrin in water ranging from about 0.09 to 0.1 µg/L. The treated mesocosms
7 contained greater numbers of fish (bluegills and some minnows), relative to untreated
8 mesocosms; however, the biomass of the fish was less (28-38%) than in control
9 mesocosms. The authors suggest that the larger number of fish in the treated ponds may
10 have resulted in increased competition for food. As noted below in Section 4.1.3.2
11 (amphibians), however, Hill et al. (1994) also noted increases in both the number and
12 biomass of tadpoles.

13 **4.1.3.2. Amphibians**

14 Compared with the information on fish, relatively little information is available on the
15 toxicity of lambda-cyhalothrin to amphibians. Pan and Liang (1996) report a 48-hour
16 LC₅₀ of 4 µg/L in tadpoles. This publication is in Chinese and was not translated for the
17 current Forest Service risk assessment. Based on English text used in Table 1 of this
18 paper, tadpoles of the marsh frog *Rana ridibunda* were tested. Pauli et al. (2000),
19 however, indicate that the species tested by Pan and Liang (1996) was the Indian rice frog
20 (*Rana limnocharis*). In either case, a 48-hour LC₅₀ of 4 µg/L is in the range of 96-hour
21 LC₅₀ values for tolerant species of fish (Table 10).

22
23 Hill et al. (1994) report an increase in tadpole populations in mesocosms treated with
24 lambda-cyhalothrin, relative to untreated mesocosms. As discussed in the previous
25 subsection, a similar observation was made on the number of fish in the mesocosms.
26 Unlike the case with fish, however, Hill et al. (1994) note an increase in both the number
27 and biomass of tadpoles.

28
29 While the information from Pan and Liang (1996) and Hill et al. (1994) does not include
30 substantial detail, neither study suggests that amphibians are more sensitive than fish to
31 lambda-cyhalothrin.

32
33 Other studies on amphibians (Cassano et al. 2003; Chio and Soderlund 2006) are focused
34 on mechanisms of action and do not provide information that has a substantial impact on
35 the risk characterization. These studies are not otherwise detailed in this abbreviated risk
36 assessment.

37 **4.1.3.3. Aquatic Invertebrates**

38 **4.1.3.3.1. Acute Toxicity**

39 Studies on the toxicity of lambda-cyhalothrin to aquatic invertebrates are summarized in
40 Appendix 4. Based on 48-hour LC₅₀ values of 0.36 µg/L in *Daphnia magna* and 0.006-
41 0.0093 µg/L in *Gammarus pulex*, U.S. EPA/EFED (1988) classifies lambda-cyhalothrin
42 as *very highly toxic* to aquatic invertebrates.
43

1 The LC₅₀ values in aquatic invertebrates cited by U.S. EPA/EFED (1988) along with
2 additional LC₅₀ values from the open literature are summarized in Table 11, which is a
3 ranked order of 48-hour LC₅₀ values from Appendix 4 (Table 1). Based on the available
4 data, the most sensitive species is *Hyaella azteca* (a species of freshwater shrimp) with
5 an LC₅₀ of 0.0023 µg/L, and the most tolerant species is *Crassostrea gigas* (the Pacific
6 oyster) with an LC₅₀ of >590 µg/L.
7

8 While the LC₅₀ values for aquatic invertebrates summarized in Table 11 may be viewed
9 as a continuum, it does appear that arthropods are more sensitive than other aquatic
10 invertebrates such as mollusks and flatworms. The higher tolerance of mollusks, at least
11 in acute toxicity studies, may be related to the ability of these organisms to retract into
12 shells and thus decrease functional exposure to living tissue (Schroer et al. 2004). The
13 basis for the apparent high tolerance of flatworms to lambda-cyhalothrin is not clear.
14 Although Schroer et al. (2004) do not provide a detailed description of the bioassays in
15 flatworms, it does not appear that these organisms were tested in a sediment-water
16 system, which would be expected to reduce the apparent toxicity of lambda-cyhalothrin,
17 based on nominal concentrations.
18

19 The data from Table 11 are illustrated in Figure 6 by segregating the toxicity data by
20 aquatic arthropods and non-arthropods. The average (geometric mean) LC₅₀ value for
21 arthropods is about 0.08 µg/L, below the corresponding mean LC₅₀ value for fish (about
22 0.5 µg/L) by a factor of 6. As noted in Table 11, two of the toxicity values for non-
23 arthropods are specified as *greater than* (>) values. In other words, the specified
24 concentration caused less than 50% mortality. Thus, the mean value of 64 µg/L is an
25 underestimate of the LC₅₀ of lambda-cyhalothrin to non-arthropod species. Using 64
26 µg/L for comparison, the tolerance of non-arthropod aquatic invertebrates to lambda-
27 cyhalothrin is greater than that in fish by a factor of about 130 [64 µg/L ÷ 0.5 µg/L ≈
28 128.8] and greater than that in aquatic arthropods by a factor of about 800 [64 µg/L ÷
29 0.08 µg/L ≈ 799.2].
30

31 As also illustrated in Figure 6, the slope of the distribution for aquatic arthropods is
32 somewhat shallower than that for fish, suggesting a somewhat greater variability in
33 sensitivities in aquatic arthropods, relative to fish. The differences are most pronounced
34 at the low proportions—i.e., the most sensitive species. Empirically, the most sensitive
35 species of aquatic arthropod—i.e., *Hyaella azteca* with an LC₅₀ of 0.0023 µ/L—is more
36 sensitive than the most sensitive species of fish—i.e., the golden orfe with an LC₅₀ of
37 0.078 µ/L—by a factor of about 34.
38

39 As noted in Section 4.1.3.1, some studies in fish suggest that the binding of lambda-
40 cyhalothrin to sediment may reduce the toxicity of lambda-cyhalothrin, based on nominal
41 concentrations. This effect is demonstrated in much greater detail in aquatic
42 invertebrates. Smith and Lizotte (2007) assayed the apparent toxicity of lambda-
43 cyhalothrin to *Hyaella azteca* and noted a relatively strong inverse relationship between
44 nominal LC₅₀ values and dissolved organic carbon. The data from Smith and Lizotte
45 (2007) are summarized in Table 12 and illustrated in Figure 7. Smith and Lizotte (2007)
46 report an r² of 0.847 but do not give the *p*-value for the correlation. In the current Forest

1 Service risk assessment, the data are reanalyzed in EXCEL. The resulting r^2 is about the
2 same ($r^2=0.840$) as that reported in Smith and Lizotte (2007) and the correlation is highly
3 significant ($p=0.000028$). While some scatter is apparent, particularly in the range of
4 about 11 mg DOC/L, the variability in the LC₅₀ values is modest—i.e., about a factor of 4
5 [$0.0111 \mu\text{g/L} \div 0.0028 \mu\text{g/L}$]. Over the full range DOC values, the LC₅₀ values span a
6 factor of about 11 [$0.0157 \mu\text{g/L} \div 0.0014 \mu\text{g/L} \approx 11.21$]. As summarized in Appendix 4
7 (Table 1), a similar dependence of the toxicity of lambda-cyhalothrin on DOC is
8 demonstrated by Day (1991) using *Daphnia magna*. As also summarized in Appendix 4,
9 Maund et al. (1998) demonstrate a greater than 100-fold decrease in lambda-cyhalothrin
10 toxicity to *Daphnia magna* in sediment/water systems, as opposed to water only systems.
11

12 Other types of acute toxicity information on lambda-cyhalothrin are available but do not
13 substantially impact the risk assessment. Heckmann et al. (2005) report behavioral
14 changes (pre-copulatory) in *Gammarus pulex*; however, the concentrations associated
15 with this effect are higher than the EC₅₀ values in sensitive species of aquatic
16 invertebrates. As with fish, some information is available on the relative toxicities of
17 lambda-cyhalothrin (two enantiomers) and gamma-cyhalothrin (one enantiomer) to
18 aquatic invertebrates. As would be expected, these studies indicate that gamma-
19 cyhalothrin is about twice as toxic as lambda-cyhalothrin (Van Wijngaarden et al. 2009;
20 Wang et al. 2007). Resistance to lambda-cyhalothrin was observed in one aquatic insect,
21 *Culex pipiens* (Lawler et al. 2007); however, the magnitude of the resistance (i.e., a factor
22 of 30) was much less than that observed in terrestrial insects (factors of up to 11,000 as
23 discussed in Section 4.1.2.4.). Maund et al. (2001) conducted an analysis of
24 concentration-time relationships in *Hyaella azteca*, noting that a log-log plot of LC₅₀
25 values and exposure time is linear—i.e., the concentration-duration relationship follows
26 Haber's Law (SERA 2007a, Section 3.3.2).

27 **4.1.3.3.2. Chronic Toxicity**

28 Information on the longer-term toxicity studies on lambda-cyhalothrin is summarized in
29 Appendix 4 (Table 2). The reproduction studies involving the exposure of *Daphnia*
30 *magna* to lambda-cyhalothrin include Barata et al. (2006), Hamer et al. (1985b), and U.S.
31 EPA/EFED (1994a,b). Most standard reproduction studies in *Daphnia magna* are
32 conducted over a 21-day period of exposure. The study by Barata et al. (2006), which is
33 also summarized in Barata et al. (2007), is an abbreviated study conducted over a 10-day
34 period of exposure. The NOEC from this study is $0.045 \mu\text{g/L}$. Apparently, at least two
35 standard 21-day reproduction studies were submitted to the EPA in support of the
36 registration of lambda-cyhalothrin. A DER is available for the early reproduction study
37 by Hamer et al. (1985b). The DER for this study, however, does not provide a detailed
38 discussion or tabular summaries of the data. Based on the discussion of this study in U.S.
39 EPA/EFED (1988), the NOEC for reproduction, $0.0085 \mu\text{g/L}$, is the most sensitive
40 endpoint. The DER for this study cites numerous deficiencies and classifies the study as
41 *Invalid* and *scientifically unsound*. Hand-written notes in the DER as well as the
42 discussion of the study in U.S. EPA/EFED (1988) indicate that the study was upgraded to
43 *Supplemental*. Apparently, a later study was submitted to the EPA, which is discussed
44 very briefly in two summary risk assessments conducted by the EPA (U.S. EPA/EFED
45 1994a,b). The study reports a reproductive NOEC of $0.00198 \mu\text{g/L}$ with a corresponding

1 LOEC of 0.0035 µg/L. In a publication on a probabilistic risk assessment of lambda-
2 cyhalothrin, Maund et al. (1998) report a 21-day reproduction NOEC of 0.002 µg/L but
3 do not provide a reference for the reported NOEC. While somewhat speculative, it
4 appears that Maund et al. (1998) are referring to and rounding the NOEC of 0.00198
5 µg/L reported in U.S. EPA/EFED (1994a,b).

6
7 There is one chronic reproduction study involving the exposure of mysid shrimp to
8 lambda-cyhalothrin. This study is summarized in U.S. EPA/EFED (1988); in addition,
9 there is a cleared review for this study (Thompson 1987). While some concerns with this
10 study are discussed in the cleared review, the study is well-described in the cleared
11 review and is classified as *Supplemental*. The NOEC for reproduction is estimated at
12 0.00022 µg/L, lower than the reproduction NOEC in *Daphnia* (0.002 µg/L) by a factor of
13 10. The NOEC for survival was 0.0017 µg/L, which is very similar to the reproductive
14 NOEC in *Daphnia*.

15 **4.1.3.3.3. Mesocosm Studies**

16 Several mesocosm studies address the effects of lambda-cyhalothrin on aquatic
17 invertebrates (Appendix 4, Table 3). Mesocosm studies basically involve exposures to
18 relatively small systems of organisms and focus primarily on detecting functional
19 changes in the system rather than impacts on the most sensitive species. The design and
20 interpretation of mesocosm studies is complex (e.g., Graney et al. 1994). In the context
21 of risk assessment, mesocosm studies are intended to provide a more *refined* or realistic
22 understanding of the environmental impact of exposure, particularly in terms of recovery.

23
24 While it is beyond the scope of the current abbreviated risk assessment of lambda-
25 cyhalothrin to review and critique each of the available mesocosm studies in detail, the
26 interpretation of the studies is reasonably consistent. Most studies that assay impacts
27 based on concentrations of lambda-cyhalothrin in water indicate transient LOEC values
28 ranging from 0.001 to 0.1 µg/L (Hill et al. 1994; Heckmann and Friberg 2005; Lauridsen
29 and Friberg 2005; Roessink et al. 2005; Schroer et al. 2004; Van Wijngaarden et al.
30 2006). This range of LOEC values spans a factor of 100, less than the variability in acute
31 EC₅₀ values for aquatic arthropods—i.e., a factor of over 1400 (Table 11). In addition,
32 the range of LOEC values is consistent with the magnitude of the 48-hour LC₅₀ values for
33 aquatic arthropods—i.e., 0.0023 µg/L for the most sensitive species and a mean LC₅₀ in
34 aquatic arthropods of 0.08 µg/L, as discussed in Section 4.1.3.3.1.

35
36 Remarkably, most of the mesocosm studies report substantial decreases in the
37 concentration of lambda-cyhalothrin in the water column over time. This decrease in
38 lambda-cyhalothrin concentrations is to be expected based on the relatively short half-life
39 of lambda-cyhalothrin in water (Section 3.2.3.4.3) as well as the impact of aquatic
40 macrophytes on the persistence of lambda-cyhalothrin in water (Section 3.2.3.4.6). The
41 decrease in lambda-cyhalothrin in water is also likely to be a factor in the reports of
42 recovery of mesocosms over time.

43
44 While the basic conclusions from the mesocosm studies are reasonably concordant with
45 laboratory bioassays, the study results are subject to interpretation. For example,

1 Heckmann and Friberg (2005) suggest that 0.5-1.0 µg/L could be regarded as a
2 community NOEC, because changes in community structure were noted only at
3 concentrations of 5 and 10 µg/L. This argument has merit in that the ecological role of
4 some sensitive species may be filled by more tolerant species. Thus, the overall structure
5 or integrity of the ecosystem is not impaired. On the other hand, it may be argued that
6 any impact of a pesticide, transient or otherwise, on any nontarget species is an
7 unintended and adverse effect. These two positions are not contradictory; they simply
8 reflect different views of the same set of observations.
9

10 As with all Forest Service risk assessments, the quantitative risk characterization for
11 aquatic invertebrates is based on ranges of toxicity values for both sensitive and tolerant
12 species (Section 4.3.3.3). Nonetheless, observations from mesocosm studies provide
13 useful information on the plausible consequence of the contamination of water with
14 lambda-cyhalothrin, and the mesocosm studies are considered further in the risk
15 characterization for aquatic invertebrates (Section 4.4.3.3).

16 **4.1.3.4. Aquatic Plants**

17 The available information suggests that aquatic plants, macrophytes and algae, are
18 unaffected by lambda-cyhalothrin at concentrations that exceed those expected after
19 normal uses of lambda-cyhalothrin. As discussed in Section 3.2.3.4.6, several studies are
20 available indicating that aquatic macrophytes play a role in rapidly absorbing and
21 metabolizing lambda-cyhalothrin. None of these studies reports adverse effects on
22 aquatic macrophytes associated with exposure to lambda-cyhalothrin. In addition,
23 several of the mesocosm studies on aquatic invertebrates (Appendix 4, Table 3)
24 specifically note the lack of adverse effects on aquatic plants at concentrations of up to
25 0.25 µg/L (e.g., Van Wijngaarden et al. 2006). Finally, Maund et al. (1998) summarize a
26 bioassay in *Selenastrum capricornutum* indicating an EC₅₀ for growth inhibition of
27 >1,000 µg/L. This toxicity value appears to be the maximum nominal concentration from
28 the study by Thompson and Williams (1985a) submitted to the U.S. EPA in support of
29 the registration of lambda-cyhalothrin in which an EC₅₀ of >310 µg/L (based on
30 measured concentrations) is reported for growth inhibition.
31

1 **4.2. EXPOSURE ASSESSMENT**

2 **4.2.1. Overview**

3 As in the human health risk assessment, all exposure scenarios for nontarget species are
4 detailed in the EXCEL workbooks that accompany this risk assessment—i.e.,
5 Attachment 1 (Chico site) and Attachment 2 (Foresthill Site)—and all exposures are
6 based on the assumption of six applications conducted at 2-week intervals with an
7 application rate of 0.08 lb a.i./acre. The exposure estimates for the various groups of
8 organisms considered in the risk assessments are summarized in the EXCEL workbooks:
9 terrestrial vertebrates (G01), honeybees (G02b), aquatic organisms (G03), herbivorous or
10 predatory insects (G08a), and soil invertebrates (G09a-c). Details of the exposure
11 assessments are discussed in the following subsections.

12 **4.2.2. Mammals and Birds**

13 **4.2.2.1. Direct Spray**

14 The unintentional direct spray of wildlife during broadcast applications of a pesticide is a
15 plausible exposure scenario. This scenario is similar to the accidental (and less plausible)
16 exposure scenarios for the general public discussed in Section 3.2.3.2. In a scenario
17 involving exposure to direct spray, the amount of pesticide absorbed depends on the
18 application rate, the surface area of the organism, and the rate of absorption.

19
20 For this risk assessment, two direct spray or broadcast exposure assessments are
21 conducted (Worksheets F01, F02). The first spray scenario (detailed in Worksheet F01)
22 assumes that a mammal weighing 20 g is sprayed directly over one half of the body
23 surface as the chemical is being applied. This exposure assessment further assumes first-
24 order dermal absorption. In an effort to encompass the increased exposure due to
25 grooming, the second exposure assessment (detailed in Worksheet F02) assumes
26 complete absorption over day 1 of exposure.

27
28 There are no exposure assessments for the direct spray of large mammals, principally
29 because allometric relationships dictate that the amount of a compound to which large
30 mammals will be exposed, based on body weight, as a result of direct spray is less than
31 the amount to which smaller mammals will be exposed, based on body weight.

32 **4.2.2.2. Dermal Contact with Contaminated Vegetation**

33 As discussed in the human health risk assessment (Section 3.2.3.3), one approach for
34 estimating the potential significance of dermal contact with contaminated vegetation is to
35 assume a relationship between the application rate and dislodgeable foliar residue.

36 Unlike the human health risk assessment, in which transfer rates for humans are
37 available, there are no transfer rates available for wildlife species. Wildlife species are
38 more likely than humans to spend long periods of time in contact with contaminated
39 vegetation. It is reasonable to assume that for prolonged exposures, equilibrium may be
40 reached between pesticide levels on the skin, rates of dermal absorption, and pesticide
41 levels on contaminated vegetation. Since, data regarding the kinetics of this process in

1 birds and mammals are not available, a quantitative assessment for this exposure scenario
2 cannot be made in the ecological risk assessment.

3 ***4.2.2.3. Ingestion of Contaminated Vegetation or Prey***

4 Separate exposure assessments are developed for acute and chronic exposure scenarios
5 involving a small mammal (Worksheets F03a, F03b, F04a and F04b), a large mammal
6 (Worksheets F10, F11a, and F11b), and large birds (Worksheets F12, F13a, and F13b).
7 Similarly, the consumption of contaminated insects is modeled for a small bird
8 (Worksheet 14a) and a small mammal (Worksheet 14b). As detailed in the exposure
9 assessment for human health (Section 3.2.3.3), the empirical relationships based on those
10 recommended by Fletcher et al. (1994) are used to estimate residues in contaminated
11 insects (Worksheets F14a and F14b).

12
13 A similar set of scenarios is provided for the consumption of small mammals by either a
14 predatory mammal (Worksheet 16a) or a predatory bird (Worksheet 16a). In addition to
15 the consumption of contaminated vegetation, insects, and other terrestrial prey, exposure
16 pathways for lambda-cyhalothrin may be associated with ambient water and fish. Thus, a
17 separate scenario is developed for the consumption of contaminated fish by a predatory
18 bird, involving acute exposure (Worksheet F08) and chronic exposure (Worksheet F09).

19
20 As discussed in Section 3.2.3.6, the exposure assessment for the general public at the
21 Chico site is assumed to involve the consumption of blackberries that grow along the
22 banks of Comanche Creek. The blackberry bushes are 50 feet from the nearest location
23 where applications of lambda-cyhalothrin will be made; accordingly, pesticide drift rather
24 than direct spray is considered. Since the consumption of blackberries may also be a
25 plausible scenario for wildlife, the possibility that some species of wildlife might
26 consume contaminated vegetation at or very close to the application site cannot be
27 excluded. Consequently, in the ecological risk assessment, the direct spray of vegetation
28 is considered for both the Chico and Foresthill sites. This is the standard and
29 conservative assumption used in most Forest Service risk assessments.

30 ***4.2.2.4. Ingestion of Contaminated Water***

31 The methods for estimating lambda-cyhalothrin concentrations in water are identical to
32 those used in the human health risk assessment (Section 3.2.3.4.6). The only major
33 differences in the estimates of exposure involve the weight of the animal and the amount
34 of water consumption. These differences are documented in the worksheets for the
35 consumption of contaminated water (F05, F06, and F07).

36
37 Estimates of the ranges of water consumption by nontarget mammals and birds are not
38 available, as they are for humans. Thus, for the acute exposure scenario, the only factors
39 affecting the estimation of the ingested dose are the field dilution rates (i.e., the chemical
40 concentration in the spilled solution) and the amount of solution spilled. In the exposure
41 scenario involving ponds or streams contaminated by runoff or percolation, the factors
42 affecting the variability of exposure levels are the water contamination rates (Section
43 3.2.3.4.2) and the application rate.

44

1 As with the human health risk assessment, the estimated water concentrations of lambda-
2 cyhalothrin are different for the Chico and Foresthill sites (Table 6). The site-specific
3 considerations regarding the plausibility of exposure during application periods—i.e.,
4 when neither creek at the Chico and Foresthill sites will be flowing—are identical to
5 those discussed in the human health risk assessment (Section 3.2.3.4.3.2).

6 **4.2.3. Terrestrial Invertebrates**

7 **4.2.3.1. Direct Spray and Drift**

8 Risks to terrestrial invertebrates following the broadcast application of a pesticide are
9 typically characterized in Forest Service risk assessments as direct deposition on the
10 insect at the nominal application rate.

11
12 This scenario is not applicable to the Foresthill site. As discussed in Section 2.3.2,
13 applications at the Foresthill site involve high-volume, high-pressure directed sprays of
14 individual-trees. While it is possible that individual insects might be directly sprayed, the
15 amount deposited on the insect would not correspond directly to the nominal application
16 rate in units of lb a.i./acre. Thus, this scenario is not included in the workbook for the
17 Foresthill site (Attachment 2), and risks to insects following direct spray are discussed
18 qualitatively in the risk characterization (Section 4.4.2.3).

19
20 This exposure scenario may also have limited applicability at the Chico site. As noted in
21 Section 2.3.1, applications at the Chico site involve airblast applications. During
22 application, the most likely scenario involves an insect flying through the cloud of
23 lambda-cyhalothrin as it is being applied. While this scenario does not involve the same
24 type of exposure as a typical broadcast application, downwind exposures associated with
25 drift may approximate exposures that might occur during broadcast applications.
26 Consequently, this exposure scenario is included in the workbook for the Chico site
27 (Attachment 1), and the limitations regarding this exposure assessment for the Chico site
28 are discussed the risk characterization (Section 4.4.2.3).

29
30 The target species for this exposure scenario is the honeybee, and the exposure is
31 modeled as a simple physical process based on the application rate and surface area of the
32 bee. The surface area of the honeybee (1.42 cm²) is based on the algorithms suggested by
33 Humphrey and Dykes (2008) for a bee with a body length of 1.44 cm. Because this
34 scenario involves acute exposure, which can only occur during or immediately after
35 application, the level of exposure will be the same, regardless of single or multiple
36 applications. Accordingly, only a single application is considered in Worksheet G02b.

37
38 The amount of lambda-cyhalothrin deposited on a bee during or shortly after application
39 depends on how close the bee is to the application site as well as foliar interception of the
40 spray prior to deposition on the bee. In addition to drift, foliar interception of applied
41 lambda-cyhalothrin is a factor in the exposure assessment for honeybees. The impact of
42 foliar interception will vary depending on the nature of the canopy above the bee. For
43 example, in studies investigating the deposition rate of diflubenzuron in various forest
44 canopies, Wimmer et al. (1992) noted that deposition in the lower canopy, relative to the
45 upper canopy, generally ranged from about 10% (90% foliar interception in the upper

1 canopy) to 90% (10% foliar inception by the upper canopy). In Worksheet G02b, foliar
2 interception rates of 0% (no interception), 50%, and 90% are used.

3 ***4.2.3.2. Ingestion of Contaminated Vegetation or Prey***

4 Like terrestrial mammals and birds, terrestrial invertebrates may be exposed to lambda-
5 cyhalothrin through the consumption of contaminated vegetation or contaminated prey.
6 For foliar applications, estimates of residues on contaminated vegetation or prey are
7 based on estimated residue rates (i.e., mg/kg residues per lb a.i. applied) from Fletcher et
8 al. (1994), as discussed in Section 3.2.3.6 of the human health risk assessment. As
9 specified in Table 8, the residue rates are estimated for four groups of food items.

10
11 An estimate of food consumption is necessary to estimate a dose level for a foraging
12 herbivorous insect. Insect food consumption varies greatly, depending on the caloric
13 requirements in a given life stage or activity of the insect and the caloric value of the food
14 to be consumed. The derivation of consumption values for specific species, life stages,
15 activities, and food items is beyond the scope of the current analysis. Nevertheless,
16 general food consumption values, based on estimated food consumption per unit body
17 weight, are available.

18
19 Reichle et al. (1973) studied the food consumption patterns of insect herbivores in a
20 forest canopy and estimated that insect herbivores may consume vegetation at a rate of
21 about 0.6 of their body weight per day (Reichle et al. 1973, pp. 1082 to 1083). Higher
22 values (i.e., 1.28-2.22 in terms of fresh weight) are provided by Waldbauer (1968) for the
23 consumption of various types of vegetation by the tobacco hornworm (Waldbauer 1968,
24 Table II, p. 247). The current risk assessment uses food consumption factors of 1.3 (0.6
25 to 2.2) kg food /kg bw. The lower bound of 0.6 is taken from Reichle et al. (1973), and
26 the central estimate and upper bound are taken from the range of values provided by
27 Waldbauer (1968).

28
29 Details concerning estimated exposure levels for the consumption of contaminated
30 vegetation by herbivorous insects are provided in Worksheets G07a, G07b, G07c, and
31 G07d. Some of the estimated doses in Worksheets G07a-d may overestimate the actual
32 amount of lambda-cyhalothrin that an insect might ingest, based on the toxicity of
33 lambda-cyhalothrin. As discussed further in Section 4.4.2.3, some of the dose estimates
34 in these worksheets lead to high hazard quotients (i.e., the ratio of the estimated dose to
35 the acute NOEC). As the insect consumes contaminated vegetation, it may become
36 intoxicated (sicken), resulting in a decreased rate of food consumption. A decrease in
37 food consumption during a dietary bioassay is an extremely common occurrence in
38 mammalian toxicity studies.

39
40 Unlike the direct spray and drift scenarios discussed in the previous subsection, the
41 exposure scenarios for the consumption of contaminated vegetation are included for both
42 the Chico site (Attachment 1) and the Foresthill site (Attachment 2). While the types of
43 applications differ at the two sites, both can be meaningfully described in terms of lb
44 a.i./acre. For both sites, however, the applications are directed foliar rather than

1 broadcast foliar. Thus, insects consuming the treated vegetation are likely to be subject
2 to higher exposures, relative to insects consuming adjacent but untreated vegetation.
3

4 ***4.2.3.3. Contact with Contaminated Soil***

5 The Gleams-Driver simulations discussed in Section 3.2.3.4.3 provide estimated soil
6 concentrations of pesticides. These estimates are given in Table 13 for the Chico and
7 Foresthill sites. The estimated peak soil concentrations are essentially identical at both
8 sites, ranging from about 0.05 to 0.6 mg/kg soil, which is to be expected because both
9 sites reflect the same application rate to similar soils. The Gleams-Driver simulations are
10 set up to give average soil concentrations in the top 12 inches of soil as well as average
11 concentrations in the entire root zone. While this can be modified, these soil depths are
12 the default method used in Gleams-Driver simulations and were not modified prior to
13 simulations. As indicated in Table 13, however, the simulations at both sites indicated
14 that the maximum penetration of lambda-cyhalothrin into the soil column is only 4
15 inches. This shallow penetration reflects the high K_{ow} for lambda-cyhalothrin, 180,000
16 g/ml. Consequently, the soil concentrations given by Gleams-Driver for the top 12
17 inches of the soil column were adjusted upward by a factor of 3 to reflect the
18 concentrations in the top 4 inches of the soil column.
19

20 The workbooks for both the Chico and Foresthill sites include three custom worksheets
21 that summarize the soil exposures for earthworms (G09a), other soil invertebrates
22 (G09b), and soil microorganisms (G09c). This approach is taken because the available
23 toxicity studies provide information on both earthworms (relatively tolerant species),
24 other more sensitive species of soil invertebrates (Section 4.1.2.4), and soil
25 microorganisms (Section 4.1.2.6).

26 ***4.2.4. Terrestrial Plants***

27 A relatively standard set of exposure assessments for terrestrial plants is developed in
28 Forest Service risk assessments on herbicides, including scenarios for direct spray and
29 spray drift, off-site transport of the pesticide by runoff, and pesticide loss from the treated
30 site by wind erosion of soil. As discussed in Section 4.1.2.5, no information is available
31 on the toxicity of lambda-cyhalothrin to terrestrial plants. Given the use of lambda-
32 cyhalothrin—i.e., an insecticide designed to protect plants from insect damage—there is
33 no basis for asserting that lambda-cyhalothrin is likely to be phytotoxic. Hence, the
34 standard exposure scenarios for terrestrial plants are not considered in this risk
35 assessment.

36 ***4.2.5. Aquatic Organisms***

37 An assessment of the effects of lambda-cyhalothrin on aquatic organisms is based on
38 estimated water concentrations identical to those used in the human health risk
39 assessment. These values are summarized in Table 6 and discussed in Section 3.2.3.4.6.
40

41 As discussed in Section 4.1.3 (hazard identification for aquatic organisms), lambda-
42 cyhalothrin will bind to both sediment and dissolved organic carbon, which will reduce
43 the apparent toxicity of lambda-cyhalothrin when expressed in units of nominal

1 concentration. While some of the toxicity data for fish and aquatic invertebrates can be
2 used to quantify the effects of sediment binding and binding to dissolved organic carbon
3 on the toxicity lambda-cyhalothrin to aquatic organisms, these effects are applicable to
4 the specific conditions of each experiment. In addition, most of the available toxicity
5 data on lambda-cyhalothrin do not specify the levels of dissolved organic carbon;
6 however, the levels are presumed to be low. Consequently, as discussed in Section
7 3.2.3.4.3, Gleams-Driver was modified to consider sediment binding in simulations of
8 streams. Thus, the water concentrations given in Table 6 reflect estimates of lambda-
9 cyhalothrin in the water column.

10
11 Similarly, the toxicity values for lambda-cyhalothrin discussed in Section 4.3.3 (dose-
12 response assessment for aquatic organisms) are presumed to reflect concentrations of
13 lambda-cyhalothrin in the water column. For the most part, this presumption is likely to
14 be correct. All toxicity values taken from studies used by U.S. EPA/OPP are based on
15 concentrations of lambda-cyhalothrin measured in the water column. The studies from
16 the open literature used in the dose-response assessment are selected because they
17 provide lower or more conservative estimates of toxicity values. Thus, the exposure
18 assessment and dose-response assessment are presumably based on comparable values—
19 i.e., concentrations of lambda-cyhalothrin in the water column.

1 4.3. DOSE-RESPONSE ASSESSMENT

2 4.3.1. Overview

3 Table 14 summarizes the toxicity values used in this risk assessment. The derivation of
4 each of these values is discussed in the following subsections. Available toxicity data
5 support separate dose-response assessments in six groups of organisms: canids, other
6 terrestrial mammals, birds, terrestrial invertebrates, fish, and aquatic invertebrates. The
7 dose-response assessment for terrestrial invertebrates is elaborated to include separate
8 toxicity values for oral and contact toxicity as well as separate toxicity values for
9 different soil invertebrates. Different units of exposure may be used for different groups
10 of organisms, depending on the nature of exposure and the way in which the toxicity data
11 are expressed. When possible, a range of toxicity values, based on the most sensitive and
12 most tolerant species within a given group of organisms, is provided.

13
14 Lambda-cyhalothrin is highly toxic to insects, particularly in terms of acute contact
15 toxicity for which the estimated NOEC is 0.0065 mg/kg bw. Lambda-cyhalothrin,
16 however, appears to be much less toxic to insects by the oral route of exposure, and the
17 estimated acute oral NOEC for lambda-cyhalothrin in terrestrial insects (0.4 mg/kg bw) is
18 only modestly lower than the NOEC in canids (0.5 mg/kg bw). The acute NOECs for
19 non-canid mammals and birds are much higher—i.e., 10 and 150 mg/kg bw, respectively.
20 Toxicity values for soil invertebrates are expressed in units of ppm of lambda-cyhalothrin
21 in soil and are not comparable to those for other groups of terrestrial organisms.
22 Earthworms appear to be relatively tolerant to lambda-cyhalothrin, with acute NOEC
23 values ranging from 10 to about 60 ppm. Other soil invertebrates, however, appear to be
24 much more sensitive to lambda-cyhalothrin. Longer-term NOECs for earthworms range
25 from 3.2 to 10 ppm, and longer-term NOECs for soil microorganisms range from 1 to 20
26 ppm.

27
28
29 Longer-term toxicity values for vertebrates can be derived for canids, other terrestrial
30 mammals, and birds. As with acute exposures, canids appear to be the most sensitive
31 terrestrial vertebrate with a chronic NOEL of 0.1 mg/kg bw/day. Unlike acute exposures,
32 the toxicity value for birds (0.85 mg/kg bw/day) is somewhat lower than the toxicity
33 value for non-canid mammals (1.5 mg/kg bw/day).

34
35 Aquatic toxicity values are derived for sensitive and tolerant fish and aquatic
36 invertebrates. For fish, separate acute toxicity values (0.000004-0.0004 mg/L) and
37 longer-term toxicity values (0.0000031-0.00031 mg/L) are derived, although the
38 differences between the acute and chronic toxicity values are small. For aquatic
39 invertebrates, the available acute toxicity data do not support the explicit derivation of
40 acute toxicity values, and chronic toxicity values (0.0000002-0.00017 mg/L) are used to
41 characterize risks associated with both acute and longer-term exposures. These values
42 encompass sensitivities in aquatic arthropods. Other aquatic invertebrates appear to be
43 less sensitive than fish or aquatic arthropods to lambda-cyhalothrin.

44

1 **4.3.2. Toxicity to Terrestrial Organisms**

2 **4.3.2.1. Mammals**

3 Forest Service risk assessments typically base the dose-response assessment for
4 mammalian wildlife on the same studies used in the dose-response assessment for human
5 health effects. As discussed in Section 3.3, the EPA bases the acute and chronic RfDs on
6 a chronic study in dogs in which the short-term NOAEL was 0.5 mg/kg bw and the
7 longer-term NOAEL was 0.1 mg/kg bw/day. Both NOAELs are based on signs of
8 neurotoxicity observed at higher doses—i.e., 3.5 mg/kg bw for short-term exposures and
9 0.5 mg/kg bw/day for longer-term exposures. For the ecological risk assessment, these
10 acute and chronic NOAELs are used to characterize risks in canids.

11
12 As noted in Section 4.1.2.1, the basis for the increased sensitivity of canids is not clear.
13 The only available toxicity study involving dogs exposed to lambda-cyhalothrin is the
14 chronic study used to derive the RfDs. Data on other mammals are limited to
15 standard toxicity studies in smaller mammals—i.e., mice, rats, and rabbits. These studies
16 suggest that non-canid mammals are more tolerant than dogs to lambda-cyhalothrin.

17
18 Developmental studies are often used to derive acute RfDs. These studies are conducted
19 over relatively brief periods of exposure and can be used to define a NOAEL for acute
20 exposure. The lowest developmental NOAEL is 10 mg/kg bw/day from a developmental
21 study in rats (Section 3.1.9.1) and is used to characterize risks associated with short-term
22 exposures in non-canid mammals. The lowest chronic NOAEL for a non-canid species is
23 12.5 mg/kg bw/day from the cancer bioassay in rats (Section 3.1.10). This NOAEL,
24 however, is above the LOAEL of 5 mg/kg bw/day from the multi-generation
25 reproduction study in rats—i.e., decreased body weights in adults and offspring. As
26 discussed in Section 3.1.9.2, conflicting summaries of the multigeneration reproduction
27 studies are given in various EPA documents. The current Forest Service risk assessment
28 relies on the most recent summary from the Health Effects Division (U.S. EPA/HED
29 2002, p. 6) of the Office of Pesticide Programs, and the NOAEL of 1.5 mg/kg bw/day is
30 used as the longer-term NOAEL for non-canid mammals.

31 **4.3.2.2. Birds**

32 There are eight studies concerning the toxicity of lambda-cyhalothrin to birds: one
33 gavage LD₅₀ study, four acute dietary studies and three reproduction studies. All studies
34 were conducted on either mallards or quail and both the acute and chronic studies suggest
35 that mallards are more sensitive than quail to lambda-cyhalothrin (Section 4.1.2.2).

36
37 Typically, gavage LD₅₀ values in birds are lower than LD₅₀ values derived from acute
38 dietary studies; however, that is not the case for lambda-cyhalothrin. As noted in Section
39 4.1.2.2, the acute gavage NOEL for lambda-cyhalothrin is 3950 mg/kg bw (Roberts and
40 Fairley 1984). As also discussed in Section 4.1.2.2, the LC₅₀ from the 5-day dietary
41 study in mallards is 3948 ppm (Roberts et al. 1985b); however, food consumption data is
42 not provided in the DER for the study. Based on typical food consumption values,
43 mallards consume food in acute dietary studies at a proportion of approximately 0.3 of
44 their body weight (SERA 2007c). Thus, the dietary LC₅₀ of 3948 ppm corresponds to a

1 dose of about 1200 mg/kg bw/day, which is lower than the gavage NOEC of 3950 mg/kg
2 bw by a factor of about 3. While somewhat unusual, this pattern is consistent with
3 observations in mammals, suggesting that some vehicles may enhance the toxicity of
4 pyrethroids (Section 3.1.4).

5
6 For acute exposures, the acute dietary study in mallards is used for the dose-response
7 assessment. The acute dietary NOEC in mallards from the study by Roberts et al.
8 (1985b) is 505 ppm, corresponding to an estimated dose of about 151.5 mg/kg bw [505
9 ppm x 0.3 mg/kg bw per ppm]. This value is rounded to 150 mg/kg bw and used to
10 characterize risks to birds associated with acute exposures.

11
12 For chronic exposures, U.S. EPA/EFED (1988) uses the reproduction study in mallards
13 by Roberts et al. (1982a). As discussed in Section 4.1.2.2, the NOEC for this study is 5
14 ppm with a corresponding LOEC of 50 ppm, based on decreased egg production. The
15 DER for the later study by Beavers et al. (1989) classifies 30 ppm as an NOEC, noting
16 that in both the 15 and 30 ppm dose levels there was an increase in food consumption that
17 was not accompanied by an increase in body weight. Moreover, increased food
18 consumption was not observed at a dietary concentration of 5 ppm. In the absence of any
19 explanation for the increased food consumption without a corresponding increase in body
20 weight, an argument might be made that the effect is associated with endocrine function.
21 As discussed in Section 3.1.8, there is some basis for concern that lambda-cyhalothrin
22 may affect normal endocrine function.

23
24 While somewhat conservative, the current Forest Service risk assessment follows the
25 same approach used by U.S. EPA/EFED (1988) and uses the dietary concentration of 5
26 ppm from Roberts et al. (1982a) to assess risks to birds associated with longer-term
27 exposures. The DER for this study does provide information on both terminal body
28 weight (an average of about 1.15 kg) and terminal food consumption (about 0.19 kg/day).
29 Based on these data, a food consumption factor of 0.17 kg food/kg body weight/day is
30 used, and the 5 ppm NOEC is converted to a daily dose of 0.85 mg/kg bw/day [5 mg/kg
31 food x 0.17 kg food/kg bw].

32 ***4.3.2.3. Terrestrial Invertebrates***

33 ***4.3.2.3.1. Contact Toxicity Value (for Direct Spray)***

34 The effects of direct spray or spray drift to terrestrial insects are typically assessed using
35 the results of contact toxicity studies—i.e., studies in which the pesticide is applied by
36 pipette to the insect. As summarized in Appendix 2 (Table 1), there are several contact
37 bioassays on terrestrial invertebrates. In addition, a field study by Hearn (1985)
38 concerning the effects of aerial sprays of lambda-cyhalothrin on honeybees is useful for
39 assessing the applicability of the contact toxicity studies to the dose-response assessment.

40
41 The four contact bioassays on honeybees yield LD₅₀ estimates of 0.409 mg/kg bw
42 (Gough et al. 1984), 0.731 mg/kg bw (Pilling and Jepson 1993), 1.1 mg/kg bw (Johnson
43 et al. 2006), and 1.74 mg/kg bw (Mayer et al. 1998), for an average value of 0.995 mg/kg
44 bw. In the field study by Hearn (1985), an application rate of 0.0075 lb a.i./acre is
45 associated with mortality in about 50% of the honeybees. An application rate of 0.0075

1 lb a.i./acre is equivalent to about 0.0834 $\mu\text{g a.i./cm}^2$. The surface area of the honeybee
2 (1.42 cm^2) is based on the algorithms suggested by Humphrey and Dykes (2008) for a
3 bee with a body length of 1.44 cm. Typical body weights for worker bees range from 81
4 to 151 mg (Winston 1987, p. 54). Taking 116 mg as an average body weight, the average
5 dose to the honeys in the field study by Hearn (1985) is about 0.00103 $\mu\text{g/mg bw}$
6 [$0.0834 \mu\text{g a.i./cm}^2 \times 1.44 \text{ cm} \div 116 \text{ mg}$] or 1.03 mg/kg bw, which is very close to the
7 0.995 mg/kg bw average from the contact bioassays with lambda-cyhalothrin.

8
9 Honeybees, however, are not the most sensitive species of bees. In both laboratory and
10 field studies, Mayer et al. (1998) noted that leafcutter bees appear to be the most sensitive
11 species with a contact LD_{50} of 0.065 mg/kg bw, a factor of about 6 lower than the lowest
12 reported LD_{50} in honeybees—i.e., 0.409 mg/kg bw from the study by Gough et al. (1984).
13 Consequently, the dose-response assessment for contact toxicity is based on the LD_{50} of
14 0.065 mg/kg bw for leafcutter bees. Mayer et al. (1998) do not provide information on
15 the NOEC or the slope of the dose-response curve. In the absence of this information, the
16 LD_{50} of 0.065 mg/kg bw is divided by a factor of 10 to approximate an NOEC of 0.0065
17 mg/kg bw. This is analogous to the approach used by U.S. EPA/OPP for risk
18 characterizations for terrestrial organisms based on LD_{50} (SERA 2007a, Table 4-2).

19 **4.3.2.3.2. Oral Toxicity Value**

20 Gough et al. (1984), the only one oral toxicity honeybee study, reports an oral LD_{50} of
21 0.909 $\mu\text{g/bee}$ and a contact LD_{50} of 0.038 $\mu\text{g/bee}$. Based on these data, the oral toxicity
22 is less than the contact toxicity by a factor of about 24 [$0.909 \mu\text{g/bee} \div 0.038 \mu\text{g/bee}$
23 ≈ 23.92]. An oral toxicity study is available concerning the exposure of the tobacco
24 cutworm, a target species, to lambda-cyhalothrin (Abro et al. 1997). As discussed in
25 Section 4.1.2.4, however, this study cannot be used to calculate a dose in units of either
26 mg/kg food or mg/kg bw. Thus, Gough et al. (1984) is the only oral toxicity study that
27 can be used directly in the dose-response assessment for the consumption of lambda-
28 cyhalothrin by terrestrial invertebrates. As discussed in Section 4.1.2.4, the oral LD_{50} of
29 0.909 $\mu\text{g/bee}$ is equivalent to about 9.7 mg/kg bw.

30
31 The DER for the Gough et al. (1984) study includes dose-response data for the direct
32 contact assay but not for the oral assay. As with the approach in the dose-response
33 assessment for contact toxicity, an NOEC is estimated from the acute LD_{50} . As indicated
34 in Appendix 2 (Table 1), the NOEC based on mortality in the contact assay is about 0.01
35 $\mu\text{g/bee}$, a factor of about 4 below the LD_{50} of 0.038 $\mu\text{g/bee}$. Thus, a case can be made for
36 dividing the oral 9.7 mg/kg bw by 4 to estimate the NOEC. Conversely, concern may be
37 expressed that the honeybee may not be the most sensitive species. As discussed in the
38 previous section, the leafcutter bee appears to be more sensitive than the honeybee by
39 about a factor of 6. Thus, for the dose-response assessment for oral exposures of
40 terrestrial insects, the oral LD_{50} of 9.7 mg/kg bw is divided by 4 to approximate an
41 NOEC for the honeybee, and is also divided by a factor of 6 to reflect plausible concerns
42 for other more sensitive species. Thus, the estimated NOEC is taken as 0.4 mg/kg bw
43 [$9.7 \text{ mg/kg bw} \div 24$].

1 **4.3.2.3.3. Soil Toxicity Values**

2 As discussed in Section 4.1.2.4 and summarized in Appendix 2 (Table 1), several toxicity
3 studies are available on soil invertebrates. For the current Forest Service risk assessment,
4 separate toxicity values are derived for earthworm as well as other soil invertebrates.

5 This approach is taken because the available information suggests that earthworms are
6 relatively tolerant to lambda-cyhalothrin, relative to other soil invertebrates.

7
8 Information on the toxicity of lambda-cyhalothrin to earthworms is available from the
9 open literature publications by Frampton et al. (2006) and Garcia et al. (2008). Both of
10 studies involve a single species of earthworm, *Eisenia fetida*. The study by Garcia et al.
11 (2008) is more detailed, providing information on acute and chronic toxicity as well as
12 avoidance. The study by Frampton et al. (2006) provides only an acute LC₅₀, which is
13 somewhat higher than the highest LC₅₀ reported by Garcia et al. (2008). Consequently,
14 all toxicity values for earthworms are taken from Garcia et al. (2008). Because Garcia et
15 al. (2008) used only a single species of earthworms, sensitivities among different species
16 of earthworms cannot be assessed. Nonetheless, substantially different toxicity values
17 are reported for different soils. Unlike the case with aquatic organism (Section 4.1.3),
18 toxicity values for earthworms are not correlated strongly with the percent organic matter
19 in the soil. While the reasons for variability in toxicity among the different soils is not
20 clear, the range of acute and chronic NOECs reported by Garcia et al. (2008) are used—
21 i.e., acute NOECs of 10 and 63.2 ppm and chronic NOECs of 3.2 and 10 ppm.

22
23 For other soil invertebrates, the Ruan et al. (2009) study yields the lowest toxicity value
24 of 0.002 ppm, based on a decrease in locomotion in soil nematodes. While the
25 concentration of 0.002 ppm may be classified as a LOAEL, the effect on locomotion at
26 this concentration appears to be slight. Thus, 0.002 ppm is used as the toxicity value for
27 the most sensitive species of soil invertebrates. In the probabilistic assessment on the
28 effects of lambda-cyhalothrin on soil invertebrates, Frampton et al. (2006) cite the
29 isopod, *Porcellionides pruinosus*, as the most sensitive species with an LC₅₀ of 0.5 ppm;
30 however, Frampton et al. (2006) do not provide an NOEC value. Thus, the reported LC₅₀
31 of 0.5 ppm is divided by 10 to approximate an NOEC of 0.05 ppm.

32 **4.3.2.4. Terrestrial Plants (Macrophytes)**

33 No dose-response assessment is proposed for terrestrial plants. As discussed in Section
34 4.1.2.5, no toxicity studies were conducted on terrestrial plants, and there is no basis for
35 asserting that lambda-cyhalothrin is likely to damage terrestrial plants.

36 **4.3.2.5. Terrestrial Microorganisms**

37 Relatively little information is available on the effects of lambda-cyhalothrin on soil
38 microorganisms but the available information suggests that adverse effects are not likely
39 (Section 4.1.2.6). Both of the available studies (Cycon et al. 2006; Latif et al. 2008)
40 involve incubation periods of at least 28 days, and the only effect that might be
41 considered adverse is a transient decrease in the populations of denitrifiers observed at 20
42 ppm by Day 14 but not at Day 28 in the Cycon et al. (2006) study. Concentrations of 0.2
43 and 1 ppm were not associated with adverse effects in any soil microorganisms. Thus, as
44 summarized in Table 14, 1 ppm is taken as the NOEC for potentially sensitive groups of

1 soil microorganisms (denitrifiers), and 20 ppm is taken as the NOEC for other soil
2 microorganisms.

3 **4.3.3. Aquatic Organisms**

4 **4.3.3.1. Fish**

5 **4.3.3.1.1. Acute Toxicity Values**

6 As summarized in Table 10, the acute LC₅₀ values for fish range from 0.078 µg/L for
7 golden orfe (Maund et al. 1998) to 7.92 µg/L for channel catfish (Kumar et al. 2007).
8 The Forest Service, however, elects to use NOEC values rather than LC₅₀ values as the
9 basis for the dose-response assessment in aquatic organisms. In the absence of
10 information on NOECs, LC₅₀ values in fish are typically divided by factors of from 10 to
11 20 to approximate an NOEC. This approach is based on the EPA use of a level of
12 concern of 0.05 for endangered aquatic species (SERA 2007a, Tables 4-2 and 4-3).
13

14 The LC₅₀ values cited above for the most sensitive and most tolerant species of fish are
15 both taken from the open literature, and the publications do not provide NOECs. As
16 detailed in Appendix 3, two sets of LC₅₀ and corresponding NOEC values are available.
17 The cleared review of Hill (1985d) in the sheepshead minnow provides a 96-hour LC₅₀ of
18 0.81 µg/L with a corresponding NOEC of 0.29 µg/L for mortality and signs of toxicity,
19 which is a factor of 2.7 below the LC₅₀. The published study involving the exposure of
20 channel catfish (Kumar et al. 2007) reports a 96-hour LC₅₀ of 7.92 µg/L; moreover,
21 mortality was not significantly different from controls at 2.5 µg/L, a factor of about 3.1
22 below the LC₅₀. In a separate assay for sublethal effects, Kumar et al. (2007) report
23 sublethal signs of toxicity, including hyperactivity and increased opercular activity at 0.8
24 µg/L, a factor of about 10 below the LC₅₀.
25

26 Because of the 10-fold difference between the LC₅₀ and the LOEC of 0.8 µg/L in the
27 Kumar et al. (2007) study, the LC₅₀ values for tolerant and sensitive species of fish are
28 divided by a factor of 20 to estimate NOECs of 0.004 µg/L [0.078 µg/L ÷ 20 = 0.0039
29 µg/L] and 0.4 µg/L [7.92 µg/L ÷ 20 = 0.396 µg/L]. As discussed in the following
30 subsection, these estimated acute NOECs are modestly higher than the estimated chronic
31 NOECs for sensitive and tolerant species of fish.

32 **4.3.3.1.1. Chronic Toxicity Values**

33 As discussed in Section 4.1.3.1.2, only two longer-term studies are available, the early
34 life-stage study in sheepshead minnow (Hill et al. 1985d) and a full life-cycle study in
35 fathead minnows (Tapp et al. 1990). The 96-hour LC₅₀ values for these two species are
36 virtually identical: 0.81 µg/L for sheepshead minnow (Hill 1985d) and 0.70 µg/L for
37 fathead minnow (Maund et al. 1998).
38

39 Based on the most sensitive endpoints in chronic studies, however, these two species
40 appear to differ substantially: an NOEC of 0.25 µg/L based on decreased embryo weight
41 for sheepshead minnow and an NOEC of 0.031 µg/L based on F₀ egg production as well
42 as F₁ embryo/larval survival for fathead minnows. As detailed in Appendix 3 (Table 2),
43 the sheepshead minnow study involved only a 32-day exposure to egg and fry. The

1 fathead minnow life-cycle study, however, is analogous to a multigeneration reproduction
2 study. For effects on the F₀ hatching, the NOEC is 0.27 µg/L (very close to the NOEC
3 for sheepshead), but the 28- and 56-day survival for F₀ larvae is 0.062 µg/L, which is
4 substantially below the 32-day NOEC of 0.25 µg/L in sheepshead minnow. Thus, the
5 fathead minnow does appear to be more sensitive than the sheepshead minnow, based on
6 reasonably comparable endpoints.

7
8 When differing chronic toxicity values are available, Forest Service risk assessments
9 often use the lowest NOEC for sensitive species and the higher NOEC for tolerant
10 species. Thus, the chronic toxicity values could be based on the reported chronic NOEC
11 for the sheepshead minnow (tolerant species) and the fathead minnow (sensitive species);
12 however, this approach does not consider the broad range of acute LC₅₀ values for other
13 species of fish. As discussed in Section 4.3.3.1.1 and summarized in Table 10, the 96-
14 hour LC₅₀ values for fish range from 0.078 to 7.92 µg/L. Both the fathead minnow and
15 sheepshead minnow appear to be near the midpoint of this range, assuming a log-normal
16 distribution of tolerances.

17
18 An alternative and more conservative approach to estimating chronic toxicity values
19 could be based on differences in the acute toxicity of lambda-cyhalothrin to fish. The
20 chronic toxicity values for sensitive species of fish are based on the ratio of the lowest
21 LC₅₀ to the LC₅₀ for fathead minnows—i.e., a factor of about 0.1 [0.078 µg/L ÷ 0.7 µg/L
22 ≈ 0.1114]. Thus, the chronic value for sensitive species of fish is taken as 0.0031 µg/L
23 [0.031 µg/L x 0.1] or 0.000031 mg/L. Similarly, chronic toxicity values for tolerant
24 species of fish are based on the ratio of the highest LC₅₀ to the LC₅₀ for fathead
25 minnows—i.e., a factor of about 10 [7.92 µg/L ÷ 0.7 µg/L ≈ 11.31]. This approach
26 results in a chronic value of 0.31 µg/L [0.031 µg/L x 10] or 0.00031 mg/L for tolerant
27 species of fish.

28
29 The mesocosm studies (Appendix 3, Table 3) are not particularly useful for assessing the
30 two alternative approaches discussed above. As discussed in Section 4.1.3.1.3, the field
31 study by Lawler et al. (2003) was conducted over a relatively brief period of time and
32 reports adverse effects on the survival of mosquito fish at an estimated concentration of
33 about 8 µg/L. The mesocosm study by Hill et al. (1994) was conducted over a longer
34 period of time with bluegills and minnows at concentrations ranging from less than 0.001
35 to 0.1 µg/L. While Hill et al. (1994) report a decrease in fish biomass relative to
36 untreated control mesocosms, this effect was not concentration-related and might reflect a
37 secondary effect on aquatic invertebrates rather than a direct effect on fish. Nonetheless,
38 if 0.001 µg/L were taken as a LOEC based on reduced biomass in fish, this would
39 support the chronic values based on the LC₅₀ values for sensitive species of fish—i.e., the
40 LC₅₀ of 0.078 µg/L with the derived chronic toxicity value of 0.0031 µg/L.

41
42 The chronic toxicity values derived from the acute LC₅₀ ratios do more fully consider all
43 of the available information. Consequently, the estimated chronic NOECs based on the
44 LC₅₀ approach are rounded to one significant figure: 0.003 µg/L for sensitive species of
45 fish and 0.3 µg/L for tolerant species of fish. These values are included in Table 14 in
46 units of mg/L and are used for assessing longer-term risks in fish.

1 **4.3.3.2. Amphibians**

2 Relatively little information is available on the toxicity of lambda-cyhalothrin to
3 amphibians (Section 4.1.3.2). The available information suggests that the sensitivity of
4 amphibians to lambda-cyhalothrin may be similar to that of relatively tolerant species of
5 fish. Because of the very limited nature of the data supporting this observation, however,
6 a formal dose-response assessment for amphibians is not developed. Risks to amphibians
7 are assumed to be comparable to those for fish (Section 4.4.3.2).

8 **4.3.3.3. Aquatic Invertebrates**

9 As discussed in Section 4.1.3.3.1 and illustrated in Figure 6, aquatic arthropods are much
10 more sensitive to lambda-cyhalothrin than are other non-arthropod aquatic invertebrates.
11 While the acute toxicity data on non-arthropod invertebrates are relatively scant, relative
12 to the toxicity data on aquatic arthropods and fish, it appears that non-arthropod aquatic
13 invertebrates are not only less sensitive than aquatic arthropods to lambda-cyhalothrin,
14 but they are also substantially less sensitive than fish to lambda-cyhalothrin. Because of
15 the substantial differences in sensitivity between arthropods and non-arthropod
16 invertebrates as well as the variability in sensitivity within aquatic arthropods,
17 the current risk assessment of lambda-cyhalothrin derives separate sets of toxicity values
18 only for sensitive and tolerant aquatic arthropods as well as sensitive and tolerant aquatic
19 non-arthropod invertebrates.

20 **4.3.3.3.1. Aquatic Arthropods**

21 Most of the toxicity data on aquatic invertebrates involve 48-hour LC₅₀ values. Similar to
22 the situation with fish, most of the studies reporting LC₅₀ values in aquatic invertebrates
23 do not provide information on NOECs. As summarized in Table 11, the LC₅₀ values for
24 aquatic arthropods range from 0.0023 µg/L for *Hyaella azteca* to 3.3 µg/L for an
25 Ostracod, not otherwise identified, (Maund et al. 1998). Acute LC₅₀ values are typically
26 divided by a factor of from 10 to 20 to estimate acute NOECs. The rationale for this
27 approach is discussed in the dose-response assessment for acute toxicity values for fish
28 (Section 4.3.3.1.1).

29
30 For lambda-cyhalothrin, the selection of the factor used to estimate the NOEC from the
31 LC₅₀ must consider the chronic toxicity values for aquatic arthropods. This consideration
32 is necessary because the acute LC₅₀ values divided by 20 yields estimated acute NOECs
33 that are somewhat lower than the experimental chronic NOECs. For examples, the
34 lowest chronic NOEC is 0.00022 µg/L for mysid shrimp (Thompson 1987). As indicated
35 in Table 11, mysid shrimp are among the more sensitive species of aquatic arthropods
36 with a 48-hour LC₅₀ of 0.0041 µg/L. The ratio of this acute LC₅₀ to the chronic NOEC is
37 about 18.6 [0.0041 µg/L ÷ 0.00022 µg/L ≈ 18.6].

38
39 The only other chronic NOEC values for aquatic invertebrates are for *Daphnia magna*.
40 Both the acute and chronic data on *Daphnia magna* are variable. As summarized in
41 Table 11, the reported acute LC₅₀ values range from 0.025 to 1.04 µg/L, spanning a factor
42 of about 40. As summarized in Appendix 4 (Table 2), the chronic NOECs for
43 reproduction range from 0.00198 µg/L (U.S. EPA/EFED 1994a,b) to 0.045 µg/L (Barata
44 et al. 2006), spanning a factor of about 23. Considering all of the available acute and

1 chronic toxicity data on *Daphnia magna*, the average LC₅₀ is about 0.33 µg/L, and the
2 average reproductive NOEC is 0.0185 µg/L. The ratio of the average acute LC₅₀ to the
3 average chronic NOEC is about 17.8 [0.33 µg/L ÷ 0.0185 µg/L ≈ 17.84].
4

5 The relationship of acute LC₅₀ values to chronic NOECs in both mysid shrimp and
6 *Daphnia magna* supports a factor of about 20 for estimating an NOEC from an acute
7 LC₅₀. In both cases, however, the NOEC is a chronic NOEC for reproduction rather than
8 an acute NOEC. It may be argued that the acute NOEC should be higher than the chronic
9 NOEC. While this argument has intuitive merit, the acute toxicity data on lambda-
10 cyhalothrin do not support the derivation of separate acute NOEC values. To the
11 contrary, Heckmann et al. (2005) reports an EC₁₀ for changes in pre-copulatory behavior
12 at 0.04 µg/L but an LC₅₀ of 5.96 µg/L—i.e., a difference of a factor of nearly 150—after
13 very short-term exposures in *Gammarus pulex*. Because of the relationship of the acute
14 LC₅₀ values to the chronic NOECs for reproduction, the current Forest Service risk
15 assessment derives only chronic NOECs, which are applied to both acute and chronic
16 exposures.
17

18 The chronic NOEC for sensitive species of aquatic arthropods is straightforward. Mysid
19 shrimp are the most sensitive species, based on chronic toxicity and almost the most
20 sensitive species based on acute toxicity. Thus, the mysid NOEC of 0.00022 µg/L
21 (Thompson 1987) is rounded to 0.0002 µg/L (0.0000002 mg/L) and used to characterize
22 risks to sensitive species of aquatic arthropods.
23

24 The chronic NOEC for tolerant species of aquatic arthropods is somewhat more complex.
25 As discussed above, the chronic NOECs for reproduction in *Daphnia magna* range from
26 0.00198 to 0.045 µg/L. *Daphnia magna*, however, is not the most tolerant species of
27 aquatic arthropod. The relationship of the acute LC₅₀ to the reproductive NOEC in both
28 mysids and daphnids is consistent—i.e., about a factor of 20 below the acute LC₅₀. Thus,
29 for tolerant species, the highest acute LC₅₀ of 3.3 µg/L for seed shrimp (*Ostracoda*)
30 (Maund et al. 1998) is divided by a factor of 20, and the estimated chronic reproductive
31 NOEC is 0.165 µg/L. This value is rounded to 0.00017 mg/L in Table 14 and used to
32 characterize risks associated with longer-term exposures in tolerant species of aquatic
33 arthropods.

34 **4.3.3.3.2. Other Aquatic Invertebrates**

35 Very little information is available on the toxicity of lambda-cyhalothrin to non-
36 arthropod aquatic invertebrates. As summarized in Appendix 4 (Table 1), Schroer et al.
37 (2004) conducted bioassays on two species of snails and one species of flatworm. LC₅₀
38 values could not be determined in any of these species over the range of test
39 concentrations. The lowest reported effect level is 8.9 µg/L. At this concentration,
40 operculum closing was noted in one species of snail (*Bithynia tentaculata*). Over
41 concentrations ranging from 0.226 to about 31 µg/L, no concentration-response
42 relationships were noted in flatworms. In a bioassay on the Pacific oyster, the LC₅₀ is
43 reported as >590 µg/L (Giddings et al. 2009).
44

1 While these toxicity data are not detailed and are limited to far fewer species than the
2 data on aquatic arthropods, it is clear that non-arthropod aquatic invertebrates (at least
3 those in the mature stage) are much less sensitive than aquatic arthropods to lambda-
4 cyhalothrin. For sensitive species, the lowest effect level of 8.9 µg/L in snails (Schroer et
5 al. 2004) is used to characterize risks. Schroer et al. (2004) classify this concentration as
6 a LOEL. Nonetheless, the response appears to be an avoidance reaction, and it is not
7 clear that toxic effects were induced. Consequently, the 8.9 µg/L LOEC is divided by 10,
8 rather than 20, to estimate an NOEC of 0.89 µg/L, rounded to 0.0009 mg/L, as indicated
9 in Table 14 (the summary table for the dose-response assessment).

10
11 For tolerant species, the toxicity value of 590 µg/L for the Pacific oyster is not well
12 defined. The only information available is that the LC₅₀ is greater than 590 µg/L;
13 however, it is not clear that adverse effects were observed. Consequently, this toxicity
14 value is treated as an LOEC and divided by 10 to estimate an NOEC of 59 µg/L, which is
15 also rounded to one significant place and entered into Table 14 as 0.06 mg/L.

16
17 No chronic toxicity studies are available on non-arthropod aquatic invertebrates. While
18 adult mollusks may be very tolerant of lambda-cyhalothrin because of their ability to
19 retract into their shell, larval stages of developing mollusks do not have this capability.
20 Consequently, no attempt is made to derive longer-term toxicity values for non-arthropod
21 aquatic invertebrates, based on relationships discussed in the previous subsection on
22 aquatic arthropods. The potential risks to larval stages of aquatic mollusks are considered
23 further in the risk characterization.

24 ***4.3.3.4. Aquatic Plants***

25 As discussed in Section 4.1.3.4, there is no basis for asserting that lambda-cyhalothrin is
26 likely to cause direct damage to aquatic plants. Consequently, no formal dose-response
27 assessment is proposed for this group of organisms, and risks to aquatic plants are
28 characterized qualitatively (Section 4.4.3.5).

1 4.4. RISK CHARACTERIZATION

2 4.4.1. Overview

3 In the ecological risk assessment, as in the human health risk assessment, the quantitative
4 expression of the risk characterization is the hazard quotient (HQ), the ratio of the
5 anticipated dose or exposure to a no-observed-effect level or concentration
6 (NOEL/NOEC) using 1 as the level of concern—i.e., an HQ of ≤ 1 is below the level of
7 concern. The specific HQs discussed in this risk characterization are based on six single
8 applications of 0.08 lb a.i./acre with a 2-week interval between applications. Since
9 lambda-cyhalothrin is not used currently at the two sites in California for which this risk
10 assessment is developed, the Forest Service might consider using somewhat lower or
11 higher application rates (up to 0.16 lb a.i./acre) resulting in a cumulative annual
12 application rate of 0.5 lb a.i./acre. Although the different rates would have an impact on
13 the specific HQs, the qualitative risk characterization would not change substantially.

14
15 Lambda-cyhalothrin is an effective insecticide and is highly toxic to insects as well as
16 other terrestrial arthropods. Within the treated area, terrestrial insects will be adversely
17 affected (and probably killed) in any effective application of lambda-cyhalothrin. Insects
18 not present at the application site will be at much lower risk, although adverse effects
19 within about 100 feet downwind of the application area are plausible. Adverse effects on
20 some soil arthropods as well as some nematodes are also plausible. Adverse effects on
21 earthworms are implausible.

22
23 Acute exposures of lambda-cyhalothrin in surface water are likely to have an adverse
24 effect on some aquatic arthropods; however, there are apparently large differences in
25 sensitivity among aquatic arthropods. Although peak concentrations of lambda-
26 cyhalothrin are likely to cause substantial mortality in sensitive species of aquatic
27 arthropods, they are not likely to cause mortality or even sublethal adverse effects in
28 tolerant species of aquatic arthropods. Fish are less sensitive than are aquatic arthropods
29 to lambda-cyhalothrin. Nonetheless, the risk characterization for fish is very similar to
30 that for aquatic arthropods—i.e., adverse effects are anticipated, based on plausible peak
31 concentrations in sensitive but not tolerant species of fish. For the Foresthill site,
32 however, this risk characterization has no practical impact because fish are not found in
33 the stream at this site.

34
35 Longer-term exposure levels are not likely to have an impact on fish or tolerant species of
36 aquatic arthropods. The possibility of adverse reproductive effects in sensitive species of
37 aquatic arthropods cannot be excluded at the Chico site and are plausible at the Foresthill
38 site.

39
40 While the relatively high HQs for sensitive species of fish and aquatic arthropods raise
41 concern for downstream contamination, this concern cannot be considered quantitatively
42 in this risk characterization. As discussed at some length in Section 3.2.3.4.6
43 (Downstream Contamination), the available information on the flow velocities and flow
44 volumes of the creeks at the Chico and Foresthill sites are not sufficiently detailed to
45 permit a quantitative discussion of downstream contamination. Nonetheless, the high

1 HQs for sensitive species of fish and aquatic arthropods at the application site suggest
2 that downstream contamination has the potential to cause adverse effects in some aquatic
3 organisms.

4
5 No plausible risks to mammals, soil microorganisms, terrestrial plants, or aquatic plants
6 can be identified. Risks to birds and non-arthropod aquatic invertebrates are not likely to
7 be substantial. The only concern for non-arthropod aquatic invertebrates involves larval
8 stage mollusks or adult mollusks without shells. No data are available on these groups.
9 It is not clear that larval stage mollusks and adult mollusks without shells would be as
10 tolerant of lambda-cyhalothrin exposures as are adult stage mollusks that have shells.

11
12 Risks to amphibians cannot be characterized directly. A plausible speculation would be
13 that amphibians may display a range of sensitivities similar to those of fish.

14 **4.4.2. Terrestrial Organisms**

15 **4.4.2.1. Mammals**

16 The risk characterization for mammals does not suggest that mammals are at risk from
17 exposure to lambda-cyhalothrin in any, including the accidental, exposure scenarios.
18 This conclusion is identical to that in the EPA risk assessments for lambda-cyhalothrin
19 (U.S. EPA/EFED 1987a,b, 1988, 1994a,b).

20
21 The highest HQ for mammals is 0.7, which is the upper bound HQ for the longer-term
22 consumption of contaminated vegetation at the application site. The highest acute HQ for
23 mammals is 0.6, which is the upper bound HQ associated with the consumption of
24 contaminated grasses or contaminated insects by mammals. While canids are considered
25 the most sensitive subgroup of terrestrial mammals, the highest HQ for canids is only 0.3.
26 Risks to canids are relatively low because canids are carnivorous and they will not
27 consume substantial amounts of contaminated vegetation.

28
29 As discussed in Section 2, all of the exposure scenarios are based on six single
30 applications of 0.08 lb a.i./acre lambda-cyhalothrin with a 2-week interval between
31 applications. If the Forest Service were to elect to use an application rate of 0.16 lb
32 a.i./acre with fewer applications and a longer interval between applications, some of the
33 acute HQs would probably exceed the level of concern, but only modestly, with no
34 substantial impact on the risk characterization.

35 **4.4.2.2. Birds**

36 The risk characterization for birds is quite similar to the risk characterization for
37 mammals and suggests that birds are not likely to be at substantial risk after applications
38 of lambda-cyhalothrin. Again, this risk characterization for birds is concordant with risk
39 characterization for birds given in risk assessments conducted by the EPA (U.S.
40 EPA/EFED 1987a,b, 1988, 1994a,b).

41
42 The only HQ that exceeds the level of concern is the upper bound HQ of 1.8 associated
43 with the longer-term consumption of vegetation at the application site. This is a worst-

1 case estimate based on the upper bound of exposure and the assumption that birds will
2 feed exclusively at the application site. Note that the central estimate for this exposure
3 scenario is 0.03, below the level of concern by a factor of over 30. Similarly, the
4 scenario for the off-site consumption of contaminated vegetation—i.e., vegetation
5 contaminated by drift—yields an upper bound HQ of only 0.02, below the level of
6 concern by a factor of 50. Finally, the study by Roberts et al. (1982a) suggests that an
7 HQ of 10 could be associated with decreased egg production but not with signs of frank
8 toxicity.

9
10 The acute toxicity data on birds suggest that birds are very tolerant to acute exposures.
11 The acute exposure scenarios for birds lead to much lower HQs. The highest upper
12 bound acute HQ for birds is 0.07, below the level of concern by a factor of over 14.

13 ***4.4.2.3. Terrestrial Invertebrates***

14 Risks to terrestrial invertebrates vary with the nature of the exposure and the group of
15 terrestrial organisms exposed. Lambda-cyhalothrin is an effective insecticide and is very
16 toxic to insects when applied as a direct spray (Section 4.3.2.3.1). As summarized in
17 Worksheet G02b in Attachment 1 (Chico site), the HQ for the direct spray of an insect
18 with no foliar interception is over 1000 and the estimated dose to the insect is about 6.8
19 mg/kg bw. As discussed in Section 4.3.2.3.1, the contact LD₅₀ values for terrestrial
20 insects range from about 0.065 to about 1 mg/kg bw. Thus, if an insect is directly
21 sprayed with lambda-cyhalothrin, the insect will probably die. While the HQ is based on
22 an application rate of 0.08 lb a.i./acre, the magnitude of the HQ makes the application
23 rate inconsequential. When used at an effective application rate or anything close to an
24 effective application rate, direct spray applications of lambda-cyhalothrin will kill insects.
25 While a direct spray scenario is not included in the EXCEL workbook for the Foresthill
26 site because of the nature of the application, an incidental direct spray of an insect during
27 tree spray applications will kill the insect. As also indicated in Worksheet G02b, risks
28 associated with airblast applications of lambda-cyhalothrin will diminish substantially
29 downwind of the application site. While the extent of drift during a specific application
30 of lambda-cyhalothrin will vary according to numerous conditions at the time of
31 application, it does not appear that risks to nontarget insects will be substantial at
32 downwind distances of greater than 100 feet from the application site.

33
34 Risks to herbivorous or predatory insects are summarized in Worksheet G08b of both
35 EXCEL workbooks that accompany the current risk assessment. While the HQs are not
36 as high as those for direct spray, the upper bound HQs range from 11 to 170, depending
37 on the type of vegetation which is treated and consumed. The doses associated with these
38 HQs range from about 4 to nearly 70 mg/kg bw (Worksheet G08a). As noted in Section
39 4.3.2.3.2, the oral LD₅₀ for lambda-cyhalothrin is about 0.9 mg/kg bw. Thus, all of the
40 upper bound HQs would be associated with insect mortality. This risk characterization is
41 intuitive: Lambda-cyhalothrin is used to kill insects on treated vegetation. This risk
42 characterization for terrestrial insects is also consistent with field studies indicating that
43 lambda-cyhalothrin will have adverse effects on terrestrial insects at very low application
44 rates—i.e., 0.0022-0.0089 lb a.i./acre (Nieheff et al. 1994).

1 Risks to soil invertebrates are variable. As summarized in Worksheet G09a, the HQs for
2 earthworms are below the level of concern by factors of at least 140 for acute exposures
3 and 30 for longer-term exposures. The acute HQs for other species of soil invertebrates,
4 however, do exceed the level of concern with HQs of 3 for the *Porcellionides pruinosus*
5 (an isopod) and up to 85 for soil nematodes.

6 **4.4.2.4. Terrestrial Plants**

7 As discussed in Section 4.3.2.4, no dose-response assessment for terrestrial plants is
8 developed because hazards to terrestrial plants cannot be identified.

9 **4.4.2.5. Terrestrial Microorganisms**

10 As with earthworms, terrestrial microorganisms do not appear to be at risk from
11 applications of lambda-cyhalothrin. As summarized in Worksheet G09c, the upper
12 bound HQ for sensitive species is 0.1, below the level of concern by a factor of 10.
13 While the data on soil microorganisms is not as extensive as that for some other groups of
14 organisms, there is no basis for asserting that applications of lambda-cyhalothrin are
15 likely to harm terrestrial microorganisms.

16 **4.4.3. Aquatic Organisms**

17 **4.4.3.1. Fish**

18 **4.4.3.1.1. Accidental Exposures**

19 As with terrestrial organisms, risks may be characterized for accidental exposures,
20 expected peak exposures, and expected longer-term exposures. The accidental exposures
21 modeled in this risk assessment are based on standard scenarios used in all Forest Service
22 risk assessments and the accidental HQs for fish are identical for both the Chico and
23 Foresthill sites. In the event of an accidental spill, the HQs substantially exceed the level
24 of concern (HQ=1) for both sensitive species of fish [HQ = 9084 (1817 to 35,958)] and
25 tolerant species of fish [HQ = 91 (18 to 360)].
26

27 As noted in Section 3.2.3.4.1, however, these accidental exposure scenarios are not
28 directly applicable to either the Chico site or the Foresthill site. At both sites, the streams
29 are not likely to be flowing during the application period. At the Foresthill site, the
30 stream beds are likely to be dry during periods of application, and there are no fish in the
31 creek at the Foresthill site (Bakke 2010). At the Chico site, pools of water may be
32 present during the application periods, but the size and composition of possible fish
33 populations are not well characterized. Nonetheless, in unusual periods in which stream
34 flow does occur, an accidental spill of a large amount of lambda-cyhalothrin at either site
35 would likely have an adverse effect on fish present in water near the site of the spill.

36 **4.4.3.1.2. Peak Expected Exposures**

37 For the Foresthill site, risks to fish are not relevant because there are no fish within this
38 reach of McBride Creek at any time (Bakke 2010). Worksheet G02 of the EXCEL
39 workbook for the Foresthill site (Attachment 2) does indicate that the acute HQs for
40 sensitive species of fish exceed the level of concern—i.e., HQs = 1.9 (0.03 to 15). These

1 values are shown for the sake of transparency but, as noted above, they do not apply
2 specifically to the Foresthill site.

3
4 Risks to fish at the Chico site are evident; however, the severity of potential effects may
5 not be substantial. As discussed in Section 4.3.3.1.1, the toxicity value for sensitive
6 species of fish is 0.004 µg/L, which is derived from a 96-hour LC₅₀ of 0.078 µg/L. As
7 summarized in Table 6, the upper bound of the peak water concentration at the Chico site
8 is 2.50 x 10⁻⁵ mg/L, which is equivalent to 0.025 µg/L, below the LC₅₀ by a factor of
9 about 3 [0.078 µg/L ÷ 0.025 µg/L ≈ 3.12]. It is not clear that this peak exposure would
10 be associated with mortality in fish. Using the general approach adopted by U.S.
11 EPA/OPP (SERA 2007a, Table 4-2), the ratio of the exposure to the LC₅₀, termed the RQ
12 by U.S. EPA/OPP, is 0.32. Although this RQ would not trigger concern for acute risk
13 (RQ ≥0.5), it would trigger concern for threatened and endangered species (RQ ≥0.05).
14 Note that the more conservative value of 0.05 for threatened and endangered species is
15 essentially equivalent to the standard approach used in Forest Service risk assessments
16 for all species. A similar interpretation of risk can be based on the relatively scant data
17 on acute NOECs for mortality in fish. As discussed in Section 4.3.3.1.1, the acute
18 NOECs for fish are factors of 2.7-3.1 below the corresponding LC₅₀ values—i.e., the
19 acute NOECs are about a factor of 3 below the LC₅₀. The simple verbal interpretation of
20 acute risks to fish at the Chico site can be stated as follows: Peak concentrations at the
21 Chico site are likely to exceed concentrations that should be regarded as acceptable;
22 however, it is not clear that fish mortality would be observed at the Chico site.

23
24 The risk characterization for acute risks to fish given in the current Forest Service risk
25 assessment is not consistent with the EPA risk characterization in the risk assessment for
26 the use of cyhalothrin on turf, trees, nurseries, and ornamental gardens (U.S. EPA/EFED
27 1994b). Specifically, the U.S. EPA risk assessment states: *Based on available*
28 *information, this proposed use pattern is not likely to cause fish kills* (U.S. EPA/EFED
29 1994b, p. 4). The difference in conclusions between the EPA risk assessment and the
30 current Forest Service risk assessment reflects the lower application rates considered in
31 the EPA risk assessment (i.e., 0.015-0.030 lb a.i./acre) as well as the EPA's use of only
32 registrant-submitted studies, rather than those studies as well as the information in the
33 published literature (Appendix 3).

34
35 The above risk characterization is based on the Gleams-Driver modeling of the Chico and
36 Foresthill site. As discussed in Section 3.2.3.4.6, Information on flow rates for
37 Comanche Creek (Chico site) suggests that maximum flow rates for Comanche Creek
38 may be underestimated by a factor of about 3. This underestimate is associated with a
39 dam upstream of the treatment site on Comanche Creek. If the flow volume of
40 Comanche Creek is greater by a factor of 3 than that modeled with Gleams-Driver, then
41 the concentrations of lambda-cyhalothrin in Comanche Creek will be less than the
42 modeled concentrations by about a factor of 3. As discussed above, the acute HQs for the
43 Chico site are 1.3 (1.1-6). If adjusted downward by a factor of 3, these HQs would be 0.4
44 (0.4-2), modestly reducing the risk characterization for sensitive species of fish at the
45 Chico site. This impact is noted but not explicitly incorporated into the worksheets,

1 because the timing and the regularity of additional flow into Comanche Creek from the
2 upstream dam is not well characterized.

3 ***4.4.3.1.3. Longer-term Expected Exposures***

4 The risk characterization for fish associated with longer-term exposures to lambda-
5 cyhalothrin at the Chico and Foresthill sites is straightforward. None of the HQs for
6 longer-term exposures in sensitive species of fish exceed the level of concern at either the
7 Chico site [HQs = 0.05 (0.003-0.1)] or the Foresthill site [HQs = 0.2 (0.003-0.9)]. The
8 upper bounds of the HQs for tolerant species of fish are below the level of concern by a
9 factor of over 100 at the Foresthill site and a factor of 1000 at the Chico site.

10
11 As with the acute HQs, the chronic HQs for the Foresthill site are included only for the
12 sake of transparency. These HQs are not directly applicable to the Foresthill site because
13 there are no fish within this reach of McBride Creek at any time (Bakke 2010).

14
15 As detailed in Section 4.3.3.1.1, the current Forest Service risk assessment uses an
16 atypical and very conservative approach in the dose-response assessment for longer-term
17 effects in fish. Rather than taking the lowest available experimental chronic NOEC, the
18 lowest experimental NOEC—i.e., the NOEC of 0.031 µg/L from the full life-cycle study
19 in fathead minnows by Tapp et al. (1990)—is adjusted downwards by a factor of 10 to
20 account for concerns that the fathead minnow may not be the most sensitive species.
21 Given this conservative approach to the dose-response assessment, there is no apparent
22 basis for asserting that longer-term exposures of fish to lambda-cyhalothrin are likely to
23 result in adverse effects.

24 ***4.4.3.2. Amphibians***

25 No formal dose-response assessment is developed for amphibians (Section 4.3.3.2). An
26 LC₅₀ 4 µg/L in tadpoles is the only available toxicity value for amphibian exposure, (Pan
27 and Liang 1996) and is close to the highest LC₅₀ in fish (7.92 µg/L). Although this
28 information as well as scant observations on amphibians from mesocosm studies (Section
29 4.1.3.2) might suggest that the sensitivity of amphibians to lambda-cyhalothrin is
30 comparable to that of tolerant species of fish, the generalization is tenuous at best. Given
31 the well-documented variability in sensitivity among fish (e.g., Figure 6), it is plausible
32 that a similar variability exists in amphibians and that the one available LC₅₀ simply
33 reflects a response in a tolerant species of amphibians. Consequently, no quantitative risk
34 characterization for amphibians is developed. In the absence of more detailed data on
35 amphibians, it seems reasonable to assume that the risk characterization for amphibians
36 will be similar to that for tolerant and sensitive species of fish (Section 4.4.3.1).

37 ***4.4.3.3. Aquatic Invertebrates***

38 ***4.4.3.3.1 Aquatic Arthropods***

39 As illustrated in Figure 6, aquatic arthropods are more sensitive than fish to lambda-
40 cyhalothrin. Consequently, risks to aquatic arthropods are similar to, albeit somewhat
41 more severe than, the risks to fish. The accidental exposure scenarios require no
42 interpretation. If a relatively large amount of lambda-cyhalothrin is spilled into a

1 relatively small body of surface water, aquatic arthropods will be killed. This is similar
2 to the risk characterization for the direct spray of a terrestrial insect.

3
4 Risks associated with expected peak concentrations of lambda-cyhalothrin in streams at
5 the Chico and Foresthill sites suggest that adverse effects are likely in sensitive but not
6 tolerant species of aquatic arthropods.

7
8 For tolerant species of aquatic arthropods, the highest HQ is 0.3, the upper bound of the
9 HQs at the Foresthill site. As discussed in the dose-response assessment (Section
10 4.3.3.3.1), the toxicity values for both sensitive and tolerant species are based on the
11 chronic NOEC. In the absence of acute NOECs, the acute LC₅₀ would typically be
12 divided by 20 to approximate an acute NOEC. This approach is not taken for lambda-
13 cyhalothrin because the acute LC₅₀ values are less than a factor of 20 above the chronic
14 NOECs and it make no sense to use an acute NOEC that is lower than a chronic NOEC.
15 This matter only modestly complicates the risk characterization. The HQ of 0.3 is
16 associated with a concentration of about 0.06 µg/L, and the highest acute LC₅₀ is 3.3
17 µg/L. In other words, the peak exposure is about a factor of 55 below the LC₅₀ for the
18 most tolerant species [$3.3 \mu\text{g/L} \div 0.06 \mu\text{g/L} = 55$]. Thus, even at the highest HQ for
19 tolerant species, substantial mortality is an unlikely effect of exposure.

20
21 For sensitive species of aquatic arthropods, however, the HQs are much higher—i.e., 27
22 (22-125) at the Chico site and 39 (0.6-295) at the Foresthill site. Except for the lower
23 bound at the Foresthill site, all of the other HQs are above 20. An HQ of 20 is associated
24 with a concentration of about 0.004 µg/L. This concentration exceeds the LC₅₀ for the
25 most sensitive aquatic arthropod—i.e., 0.0023 µg/L for freshwater shrimp reported by
26 Maund et al. (1998). Thus, expected peak concentrations of lambda-cyhalothrin are
27 likely to cause mortality in sensitive species of aquatic arthropods. This conclusion
28 applies at the Chico site and in most years at the Foresthill site.

29
30 The risk characterization for longer-term exposures at the Chico and Foresthill sites are
31 similar for tolerant species of aquatic arthropods. The highest HQ is 0.003, the upper
32 bound of the HQ at the Foresthill site, which is below the level of concern by a factor of
33 over 300. The upper bound HQ at the Chico site is 0.002, below the level of concern by a
34 factor of 500. Thus, there is no plausible basis for asserting that longer-term exposures to
35 lambda-cyhalothrin are likely to pose a risk to tolerant species of aquatic arthropods at
36 either the Chico or Foresthill sites.

37
38 The longer-term HQs for sensitive species of aquatic arthropods are also similar at both
39 sites—i.e., 0.8 (0.5-1.5) at the Chico site and 0.4 (0.02-2) at the Foresthill site. As
40 discussed in Section 4.3.3.3.1, the chronic NOEC for lambda-cyhalothrin is 0.00022 µg/L
41 in mysid shrimp (Thompson 1987). The LOEC in this study is 0.00046 µg/L, based on a
42 decrease in the number of young, and is equivalent to an HQ of about 2 [$0.00046 \mu\text{g/L} \div$
43 $0.00022 \mu\text{g/L} \approx 2.091$]. Thus, although risks associated with the upper bound HQ of 1.5
44 at Chico site cannot be clearly interpreted, the possibility of adverse reproductive effects
45 in sensitive species of aquatic arthropods cannot be excluded at the Chico site and are
46 plausible at the Foresthill site.

1
2 While the risk characterization for aquatic arthropods is somewhat complex, this risk
3 characterization is similar to the risk characterization that could be derived from
4 mesocosm studies. As discussed in Section 4.1.3.3.3, most mesocosm studies suggest
5 that adverse effects in sensitive species of aquatic invertebrates are plausible and that
6 species composition may be affected. Nonetheless, the overall structure and functioning
7 of the ecosystem may not be substantially impaired, and recovery is likely to be observed
8 as concentrations of lambda-cyhalothrin decline. While the current Forest Service risk
9 assessment focuses on effects in sensitive species, the overall risk characterization for
10 aquatic arthropods is consistent with effects reported in mesocosm studies.
11

12 As discussed in Section 4.4.3.1.2, the concentrations of lambda-cyhalothrin in Comanche
13 Creek (Chico site) may be overestimated by a factor of about 3, due to the impact of a
14 dam upstream of the Chico site. The only HQs of concern for the Chico site are the acute
15 HQs for sensitive species of aquatic arthropods—i.e., 27 (22-125). Reducing these HQs
16 by a factor of 3 would have no impact on the risk characterization.

17 **4.4.3.3.2. Other Aquatic Invertebrates**

18 The risk characterization for non-arthropod invertebrates is somewhat circumscribed by
19 the limited data on this group of organisms (Section 4.3.3.3.2). As with amphibians, the
20 data on non-arthropod aquatic invertebrates are much less extensive and detailed than the
21 data on fish and aquatic arthropods. Nonetheless, information is available on several
22 species and it seems clear that at least some groups of non-arthropod aquatic
23 invertebrates are much less sensitive than fish or aquatic arthropods to lambda-
24 cyhalothrin (Figure 6). In addition, there is a biological basis for this tolerance in mature
25 mollusks—i.e., the ability to reduce soft tissue exposure by closing or withdrawing into
26 shells. The risk characterization for this group, however, is poorly defined because the
27 likelihood and nature of adverse effects cannot be clearly articulated.
28

29 At least for mollusks, the following relatively benign risk characterization is limited to
30 adult stage organisms that have shells. This is the only subgroup of mollusks on which
31 data are available. No information is available on adult mollusks without shells as well
32 as larval stage aquatic mollusks. While somewhat speculative, small larval stage
33 mollusks might be as susceptible as small aquatic arthropods to lambda-cyhalothrin.
34 Conversely, one species of flatworm—i.e., *Polycelis nigra/tenuis* from the study by
35 Schroer et al. (2004)—is also tolerant to lambda-cyhalothrin. The basis of this tolerance
36 is unclear but might reflect a more general tolerance in non-arthropod aquatic
37 invertebrates.
38

39 As with other groups of aquatic organisms, accidental exposures lead to HQs that
40 substantially exceed the level of concern for sensitive but not tolerant species of non-
41 arthropod aquatic invertebrates—i.e., 40 (8-160) for sensitive species and 0.6 (0.1-2) for
42 tolerant species. It does not seem plausible that lethality or even sublethal toxicity would
43 be observed in some tolerant species. Even in sensitive species, the nature of possible
44 adverse effects is unclear. In the event of an accidental spill, the most likely response of
45 mollusks will be to retract into or close their shells. Avoidance might be noted in more

1 mobile benthic organisms. While more serious effects cannot be ruled out, the
2 probability of observing widespread mortality is not clear.

3
4 For expected peak concentrations, the risk characterization is reasonably clear. The
5 highest HQ is 0.07, the upper bound HQ at the Foresthill site. This HQ is below the level
6 of concern by a factor of about 14. Given the nature of the acute toxicity information, it
7 does not seem likely that this HQ would be associated with adverse effects in adult
8 mollusks. At the Chico site, the highest HQ is 0.03, below the level of concern by a
9 factor of over 30.

10
11 No data are available on the chronic toxicity of lambda-cyhalothrin to non-arthropod
12 invertebrates, and no risk characterization for chronic exposures can be developed.

13 ***4.4.3.4. Aquatic Plants***

14 While no dose-response assessment for aquatic plants is developed in the current Forest
15 Service risk assessment, there is relatively little uncertainty in the risk characterization
16 for aquatic plants. There are ample studies relating to the metabolism of lambda-
17 cyhalothrin by aquatic plants (Section 3.2.3.4.6) as well as mesocosm studies involving
18 aquatic plants (Appendix 4, Table 3) to support the assertion that lambda-cyhalothrin will
19 not damage aquatic plants at concentrations greater than or equal to those anticipated in
20 the normal use of this insecticide. However, losses in aquatic invertebrate populations
21 could result in changes in biomass and species composition of aquatic plants.

22

5. REFERENCES

NOTE: The initial entry for each reference in braces {} simply specifies how the reference is cited in the text. The final entry for each reference in brackets [] indicates the source for identifying the reference.

SET00	Papers from preliminary scoping.
SET01-Toxline01	Standard initial TOXLINE literature search.
SET01-Toxline02	Supplemental TOXLINE search on resistance - lambda-cyhalothrin and esfenvalerate only.
Set01-ATSDR	Tree search of ATSDR 2003.
SET02	Additional papers after initial screen.
SET03	Papers relating to pesticide dilution in streams.
Sec	Studies cited from secondary sources.
EPA Reviews	Cleared reviews available from http://www.epa.gov/pesticides/foia/reviews/128897/index.htm .
E-Docket01	255 files at www.regulations.gov - e.g., EPA-HQ-OPP-2007-1024-0507, EPA-HQ-OPP-2007-1024. Reviewed and selectively downloads. Several MRIDs/DERs were extracted from some of the files. These are listed separately in this bibliography.
Internet	Various publications located on the Internet.
Std	Standard references used in most Forest Service risk assessments.

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LAMBDA-CYHALOTHRIN - insecticide

2002 estimated annual agricultural use

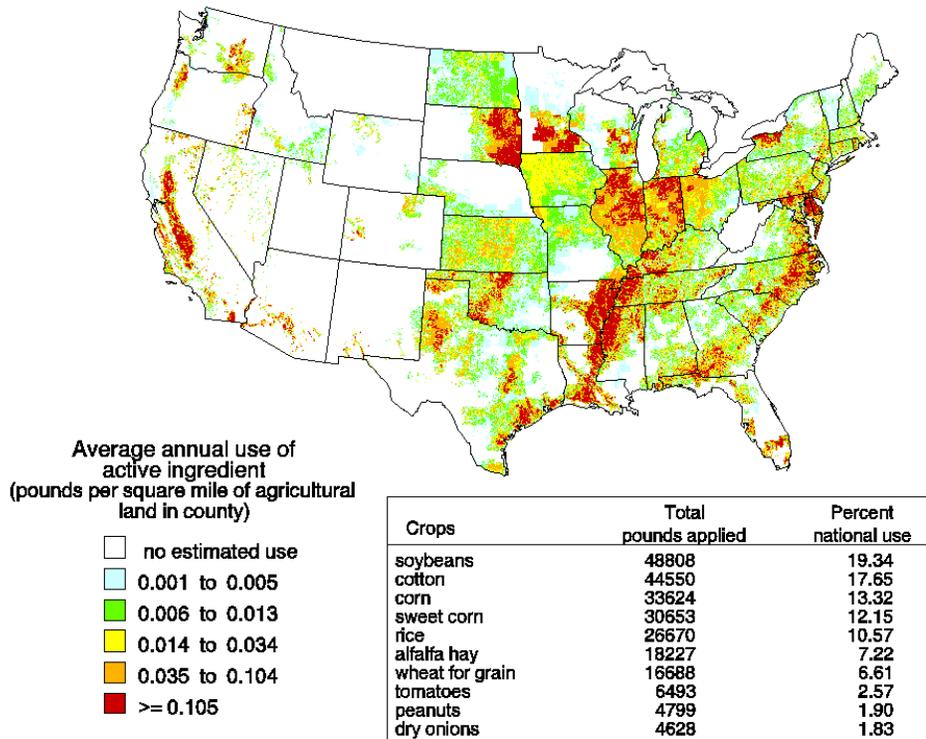


Figure 1: USGS Use Map for Lambda-Cyhalothrin During 2002

Source: USGS (2003)

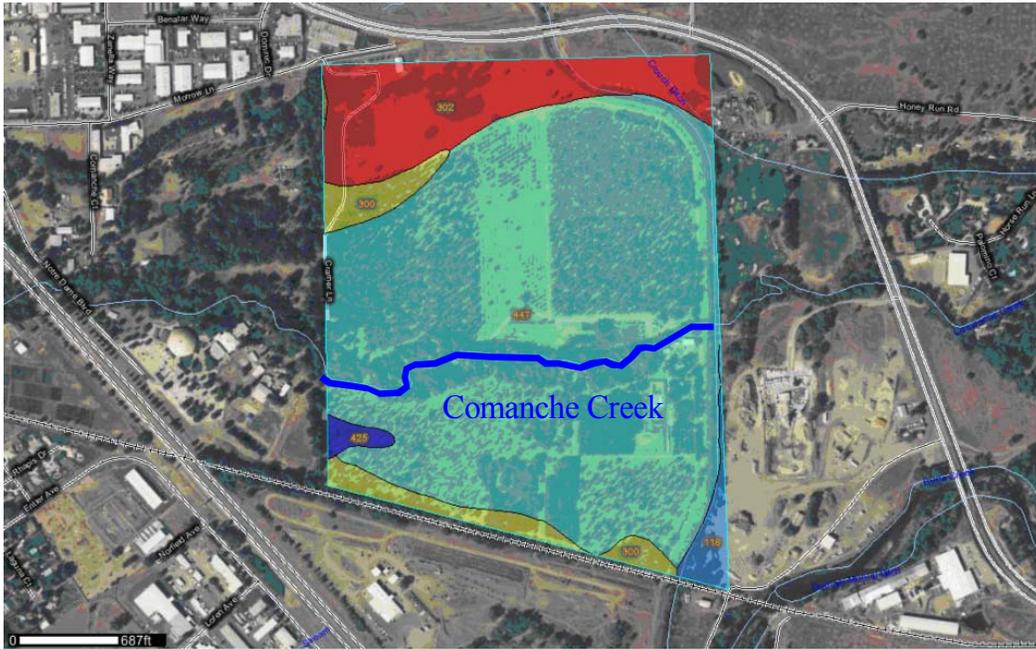
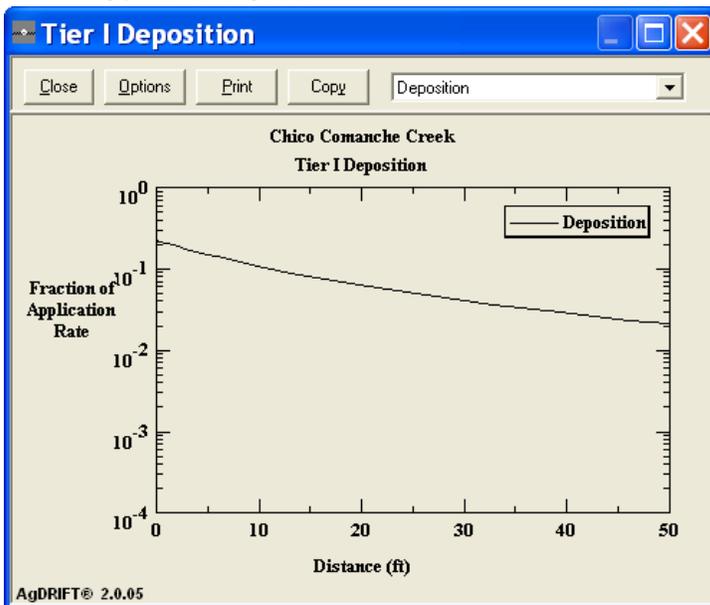


Figure 2: Chico Site Aerial View

Note: The color shaded regions in the above figure represent different soil types from the USGS soil survey in the area of interest used for the Gleams-Driver simulations. See Section 3.2.3.4.3 for discussion. Region 447 is the large area encompassing Comanche Creek.



The output to the left is from AgDrift, Version 2.0.05 using drift for orchard airblast based on “Orchard”, which gives higher drift rates than the “Normal” Orchard in AgDrift.

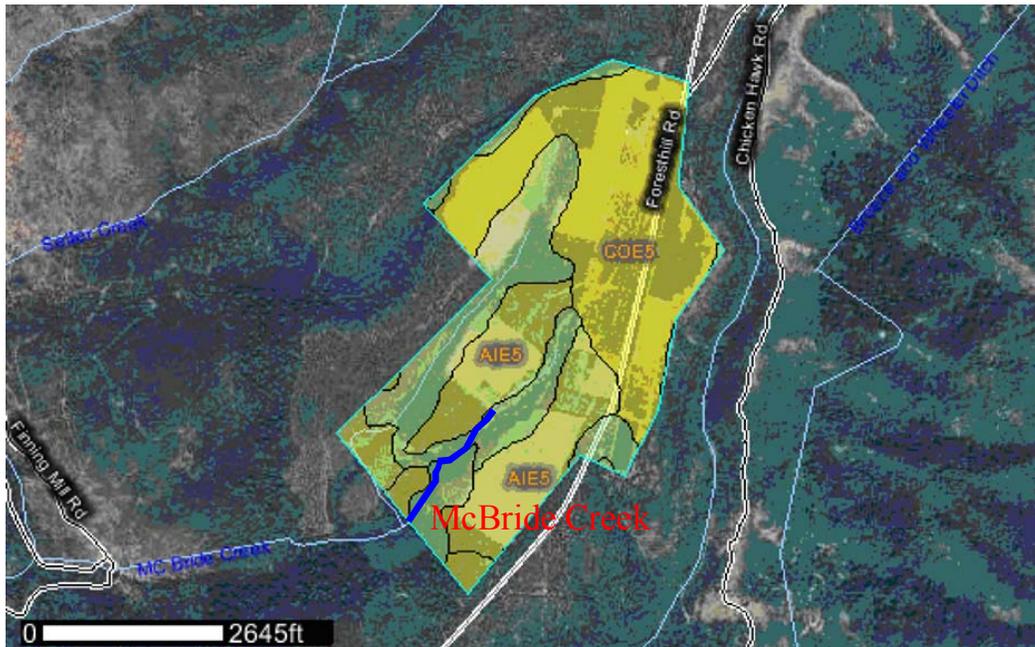
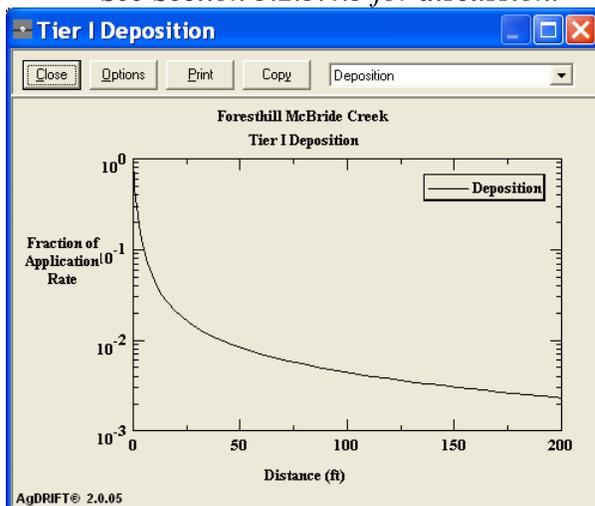


Figure 3: Foresthill Site Aerial View

Note: The color shaded regions in the above figure represent different soil types from the USGS soil survey in the area of interest used for the Gleams-Driver simulations. See Section 3.2.3.4.3 for discussion.



Above is based on 50 percentile deposition high boom ground broadcast with ASAE fine to medium coarse droplets.

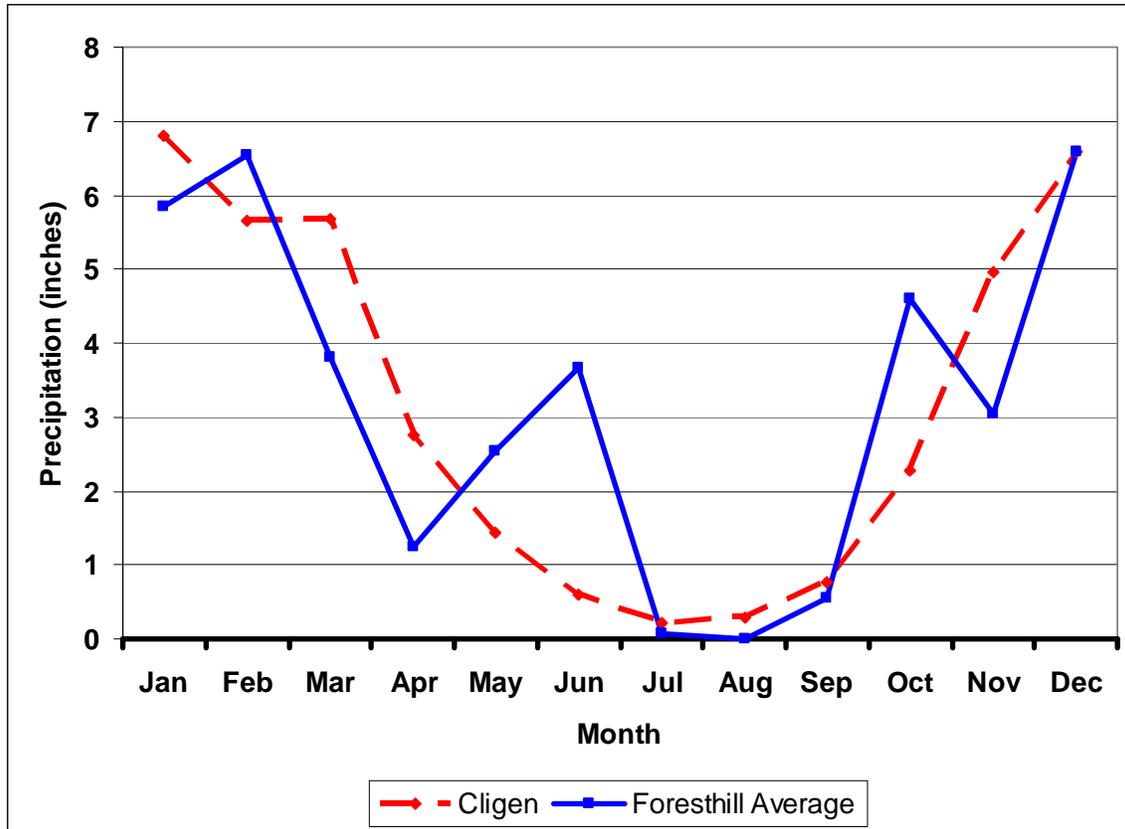


Figure 4: Foresthill Precipitation, Comparison of Cligen Simulation to Historical Data

Historical data from 2007 to 2009 taken from www.foresthillweather.com.
See Section 3.2.3.4.3.1 for discussion.

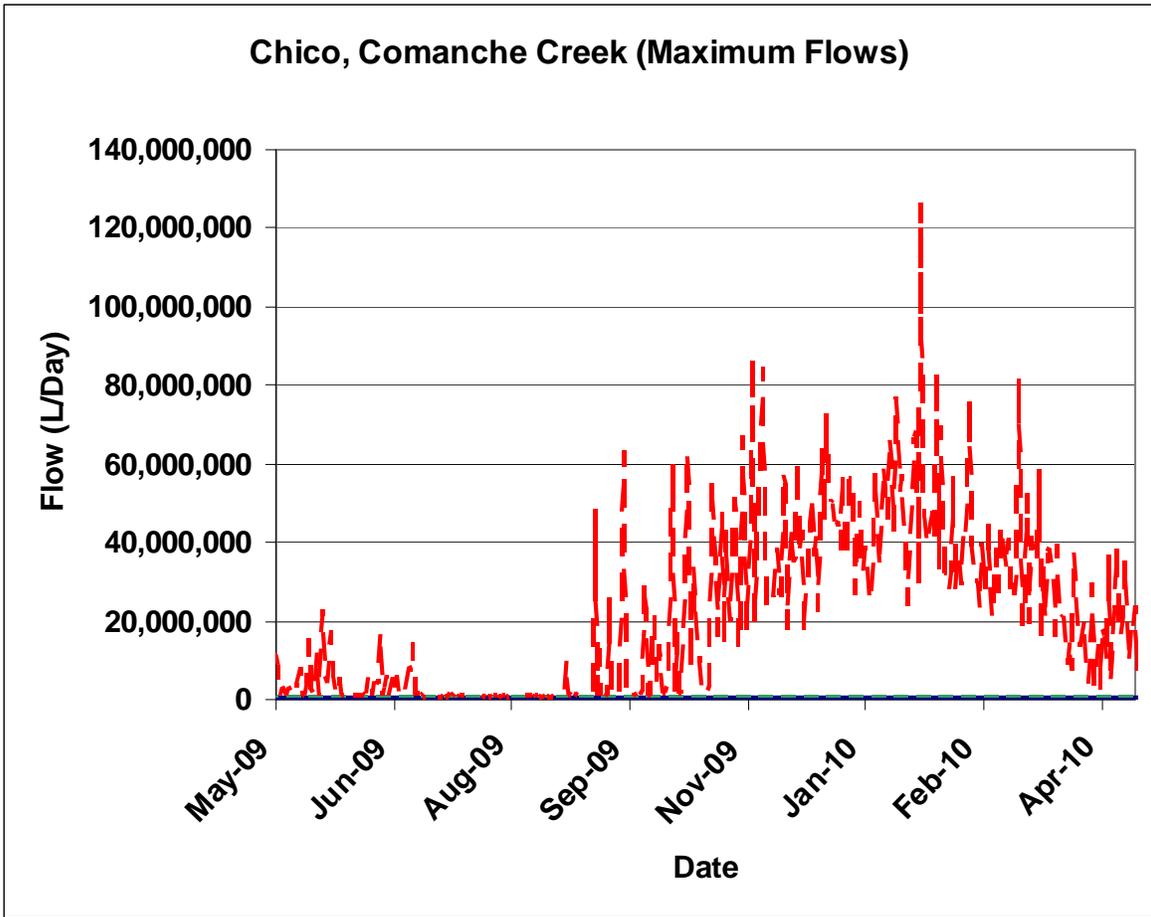


Figure 5: Daily Peak Stream Flows at Chico Site

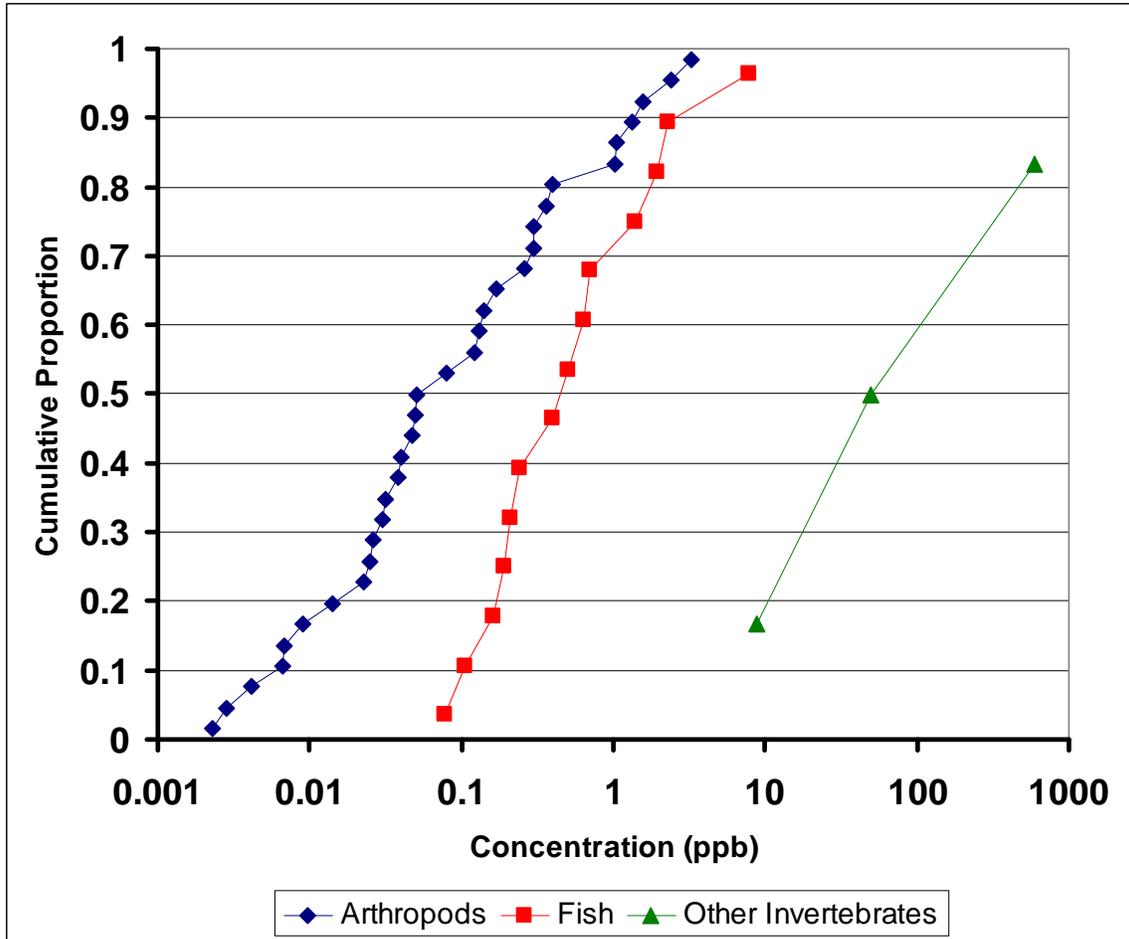


Figure 6: Acute toxicity values for aquatic animals

See Tables 10 (fish) and 11 (aquatic invertebrates) for data.
See Section 4.1.3. for discussion.

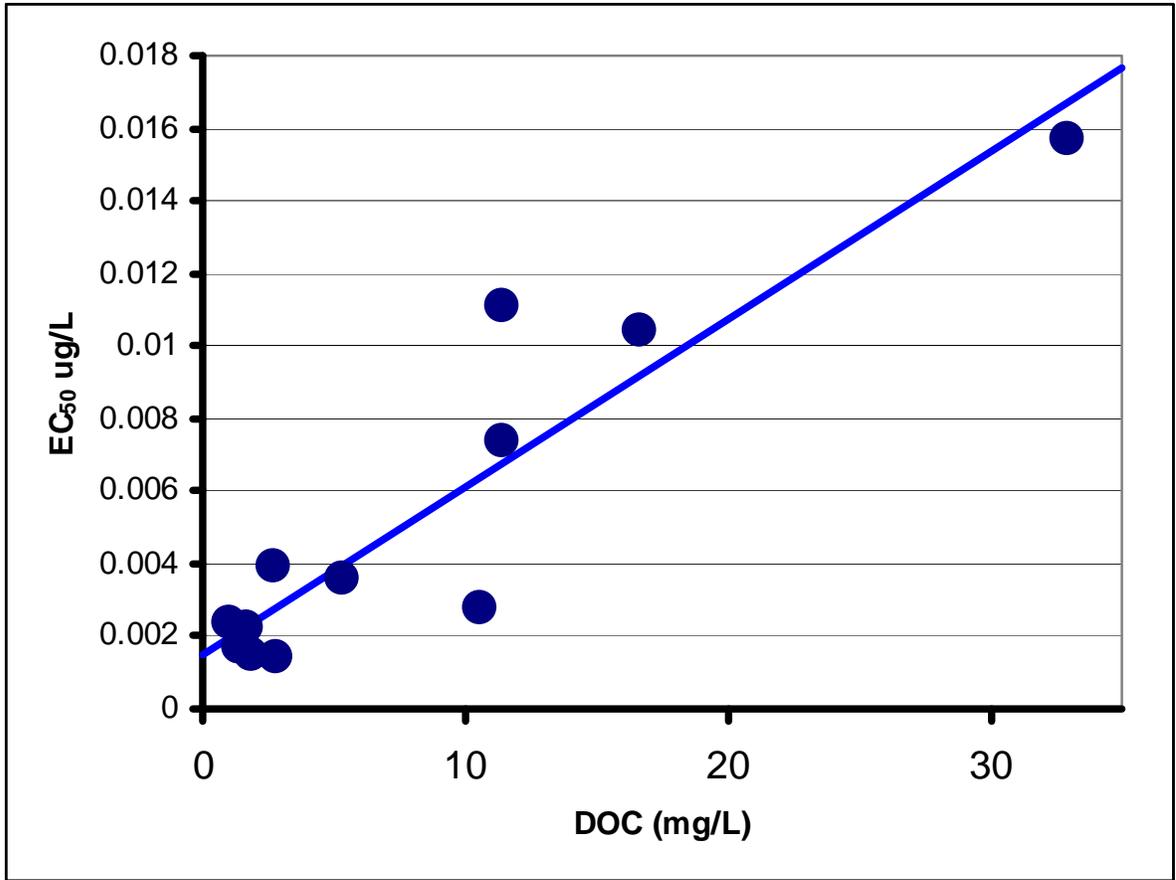


Figure 7: Impact of DOC on toxicity to *Hyaella azteca*

Data summarized in Table 12.
See Section 4.1.3.3 for discussion.

Table 1: Properties of lambda-cyhalothrin

Property	Value	Reference
Nomenclature Common Name	Lambda-cyhalothrin	Tomlin 2004
CAS Name	1a(S*),3a(Z)]-(±)-cyano(3-phenoxyphenyl)methyl 3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethylcyclopropanecarboxylate	Tomlin 2004
Appearance/state, ambient	Colorless solid; (tech. is a dark brown/green solidified melt).	Tomlin 2004
Bioconcentration	4600 to 5000	U.S. EPA/OPP 1988a U.S. EPA/EFED 1989a
CAS number	91465-08-6	Tomlin 2004
Density	1.33 g/ml (25 °C)	Tomlin 2004
Foliar half-life	5 days	Knissel and Davis 2000
Foliar washoff fraction	0.4	Knissel and Davis 2000
Fruit and vegetation halftimes	4.9 – 7 days (grapes)	Banerjee et al. 2006
	10 days (apples)	Bostanian et al. 1993
	3.6 – 4.5 days (tomatoes)	Jayakrishnan et al. 2005
	3 days (egg plant)	Mukherjee and Gopal 1992
	2.9-4.0 days (tea)	Seenivasan and Muraleedharan 2009
	2.9 days (cabbage)	Zhang et al. 2006
	3.4 days (tea)	Zongmao and Haibin 1997
Henry's law constant	$2 \times 10^{-2} \text{ Pa m}^3 \text{ mol}^{-1}$	Tomlin 2004
Kd	1,970 to 7,610	U.S. EPA/OPP 2002a
Koc (g/ml)	180,000	USDA/ARS 1995; Knissel and Davis 2000
	330,000	Tomlin 2004
	326,000	Amweg et al. 2005
log K _{ow}	7 (20 °C) [Kow = 10,000,000]	Tomlin 2004 USDA/ARS 1995 Bennett et al. 2005
Melting point	49.2 °C; (tech., 47.5-48.5 °C)	Tomlin 2004 USDA/ARS 1995
Molecular formula	C ₂₃ H ₁₉ ClF ₃ NO ₃	USDA/ARS 1995
Molecular weight (g/mole)	449.9	Tomlin 2004
Sediment-Water half life	5-11 hours	Tomlin 2004
Soil half life, field dissipation	43(30-84) days 14 to 60 days 1.3 weeks	USDA/ARS 1995 U.S. EPA/OPP 1988a Hill and Inaba 1991
Soil metabolism half life (aerobic)	23-82 days <30 days	Tomlin 2004 U.S. EPA/OPP 1988a

Property	Value	Reference
	30 days	Knissel and Davis 2000
	42.6	Bennett et al. 2005
U.S. EPA Docket Number	EPA-HQ-OPP-2007-1024-0507, EPA-HQ-OPP-2007-1024, and several more	www.regulations.gov
Vapor pressure	2×10^{-4} mPa (20 °C, est.)	Tomlin 2004; USDA/ARS 1995
Water, aquatic metabolism half life	7-15 days	Tomlin 2004
Water, dissipation half lives	0.5 – 2.7 days (water column in ditches)	Arts et al. 2006
	1 day (pond mesocosms)	Farmer et al. 1995
	0.3 – 1 day	Gu et al. 2007
	1 day	Hand et al. 2001
	1 day	Hill et al. 1994
	1 (0.7-1.2) days (freshwater mesocosm)	Van Wijngaarden et al. 2004
Water solubility (mg/L)	0.005 mg/l (20 °C)	Tomlin 2004 USDA/ARS 1995

Table 2: Selected formulations of lambda-cyhalothrin

Trade Name ^a	Supplier	EPA Reg. No. (Date of most recent EPA label)	Type of Formulation	Active Ingredient (% by weight)	Lbs a.i. per Gallon
Grizzly Z	Winfield Solutions LLC	1381-211 (3/10/08)	Liquid	11.4%	1.
Kaiso 24WG	Nufarm Americas, Inc.	228-526 (4/28/09)	Granular	24%	N/A
Lambdastar 1 CS April 9, 2004	LG Life Sciences	71532-25 (4/9/04) [Note 2]	Liquid	12%	1.
Lambda-CY EC Insecticide-RUP	United Phosphorus, Inc	70506-121 (11/10/08)	Liquid	11.4%	1.
Lambda-T	Helena Chemical Co.	100-1112-5905, 2/10/09 [Note 3]	Liquid	11.4%	1.
Silencer	Makhteshim Agan of North America	66222-104 (11/20/08)	Liquid	12.7%	1.
Taiga Z	Agriliance, LLC	1381-211 or 100-1112-1381 (3/5/07) [Note 1]	Liquid	11.4%	1.
Warrior Insecticide with Zeon Technology	Syngenta	100-1112 (2/26/09)	Liquid	11.4%	1.
Warrior II with Zeon Technology	Syngenta	100-1295 (9/23/08)	Liquid	22.8%	2.08

^a Label information from the U.S. EPA/OPP label system (<http://oaspub.epa.gov/pestlabl/ppls.home>) unless otherwise specified.

Note 1: The U.S. EPA/OPP label gives the EPA Reg. No. 1381-211 for Taiga Z, which is identical to the Reg. No. for Grizzly Z. The Agrisolutions label give the Reg. No. as 100-1112-1381. Note that 100-1112 is the EPA Reg. No. for Warrior.

Note 2: EPA Est. Nos. 5905-GA-01, 66196-CA-01.

Note 3: The U.S. EPA label system does not currently accept 100-1112-5905 designation for a label search. A specimen label for Lambda-T was obtained from <http://www.cdms.net>.

Table 3: Inerts Contained in End-use Formulations Based on MSDSs

Formulation (% a.i.) ^a	Inerts: Name, CAS No. from MSDS	Inert % by Weight
Grizzly X (11.4%) <i>Same as Taiga and Lambda-T</i>	Naphthalene	≤1.4%
	Propylene Glycol	N.S.
	Petroleum Solvent (NOS)	N.S.
	Total inerts not quantified	≥87.2%
Kaiso 24WG (24%)	N-methyl pyrrolidone (CAS 91465-08-6)	2%
	Total inerts not quantified	74%
Lambda-Cy EC (11.4%)	Solvesso 200	78%
	Total inerts not quantified	10.6%
Lambdastar 1 CS (12%)	Total inerts not quantified	88%
Lambda-T (11.4%) <i>Same as Grizzly and Taiga</i>	Naphthalene	≤1.4 %
	Propylene Glycol	N.S.
	Petroleum Solvent (NOS)	N.S.
	Total inerts not quantified	≥87.2%
Silencer (12.7%)	Aromatic solvent (CAS 64742-94-5)	78.4%
	Naphthalene (91-20-3)	7.84%
	Total inerts not quantified	1.06%
Taiga Z Insecticide (11.4%) <i>Same as Grizzly X and Lambda-T</i>	Naphthalene	≤1.4%
	Propylene Glycol	N.S.
	Petroleum Solvent (NOS)	N.S.
	Total inerts not quantified	≥87.2%
Warrior (11.4%) <i>Similar to Grizzly X, Lambda-T, and Taiga Z</i>	Naphthalene	<1.5%
	Propylene Glycol	N.S.
	Petroleum Solvent (NOS)	N.S.
	Total inerts not quantified	>87.1%
Warrior II (22.8%)	Petroleum solvent	N.S.
	Titanium dioxide	N.S.
	Total inerts not quantified	77.2%

^a Information from Material Safety Data Sheets.

Table 4: Chico and Foresthill Site Characteristics

Parameter	Chico	Foresthill
Latitude	39°42'41.32" N [39.711478]	39°05'05.34" N [39.084817]
Longitude	121°46'54.59" W [121.781831]	120°44'00.34" W [121.733428]
Elevation	323 feet	4200 feet
Weather Data	Chico Exp Station, CA	Linear interpolation
Average Annual Temperature	60.6 °F	52.0 °F
Average Annual Precipitation (inches)	25.3 inches	38.0 inches
Applications	0.08 lb a.i./acre, 6 applications every 2 weeks starting in May 0.16 lb a.i./acre, 3 applications every 4 weeks starting in May	
Soil Inputs ^b		
Soil Depth (to water table)	40 inches	80 inches [actual value >78 inches]
Soil Classification	Fine Sandy Loam	Loam
Drainage Class	Moderately well drained	Well drained
Hydrologic Soil Group	B	B [B,B,B,C]
Soil Erodibility Factor (KSOIL) ^d	0.18	0.2
Clay (%)	15.1	31.3
Silt (%)	18.0	34.7
Organic Matter (%)	1.78	2.54
Bulk Density (g/cm ³)	1.55	1.21
Porosity (cc/cc)	0.4 ^d	0.4 ^d
Field capacity(cc/cc)	0.37 ^d	0.26 ^d
Wilting Point(BR15, cc/cc)	0.12	0.155
SCS curve number (CN2)	55 ^f	59 ^f
Soil evaporation parameter (CONA)	3.5 ^d	4.5 ^d
Saturated Conductivity Below Root Zone (RC) ^h	25 µm/sec 3.5 inches/hr	2.9 µm/sec 0.41 inches/hour
Saturated Conductivity within Root Zone (SATK)	21.82 µm/sec 3.12 inches/hr	7.94 µm/sec 1.13 inches/hr
Manning's "n"	0.01	0.01
Field Characteristics ^c		
Treated area	83.7 acres	45 acres ^j
Total Field area (for drainage)	203 acres	342 acres ^j
Longest flow path (feet)	2000 feet	1150 feet
Representative slope	1%	5% ⁱ
Depth to Restrictive Layer	>40 inches	53 inches
Stream Characteristics ^c		
Name	Comanche Creek	McBride Creek
Shortest distance from application site to stream	50 feet	200 feet
Water Flow Rate	Ephemeral due to diversion operations in winter. Maximum flow of 150 cfs. ⁱ	Ephemeral: Dry in summer to early winter
Proportional drift to stream	0.002 to 0.02	0.0002 to 0.002
Functional stream section length for drift (feet)	500	250
Average width (feet)	15 feet (≈4.6 meters)	N/A (assume 2 meters or ≈ 6 feet)

^a Data obtained at USGS National Water Information System: Web Interface, <http://waterdata.usgs.gov/nwis/nwisman>.

^b Soil inputs are based on data from the USDA Soil Survey at <http://websoilsurvey.nrcs.usda.gov/app/> where available. For Chico site, values are based on Map Unit 447 (77.1% of AOI). For Foresthill site, the values are based primarily on Map Units AIE5 (23.8% of AOI), COE5 (42.3% of AOI), and CSE5 (25.2% of AOI) using weights of 1, 2, and 1, respectively.

^c Treatment area specified by the USDA Forest Service. The field size is taken aerial imagining from <http://websoilsurvey.nrcs.usda.gov/app/>. Stream details from Bakke 2009c except as otherwise noted.

^d Default values from Knisel and Davis (2000).

^e For Chico, this is the mean of the range, 0.15 to 0.3, given by Knisel and Davis for Soil Group B. For Foresthill, this is the 3:1 weighted mean for Group B (0.15 to 0.3) and Group C (0.05 to 0.15).

^f For Chico, this is the central value for Group B for woods in good condition. For Foresthill site, this is the 3:1 weighted mean of the central values for Groups B and C for woods in good condition.

^g For Chico site, the stream is *Comanche Creek*. For the Foresthill site, the stream is *McBride Creek*.

^h Calculated from SATK in units of µm/sec from USDA Soil Survey at <http://websoilsurvey.nrcs.usda.gov/app/>. And converted to units of inches/hr for input into Gleams-Driver – 1 µm/sec = 0.141732 inches/hour.

ⁱ From Chico FEIS (USDA/FS 1998). Corresponds to about 350,000,000 L/day.

^j Post-peer review inputs from Region 5 indicated the following changes: treated area is 45 acres not 118 acres, total area of site is 342 acres and not 507 acres, the characteristic slope is 5% not 16%.

Table 5: Chemical parameters used in GLEAMS modeling.

Parameter	Clay	Loam	Sand	Note/ Reference
Halftimes (days)				
Aquatic Sediment		0.5		Note 1
Foliar		5		Knissel and Davis 2000
Soil		30		Note 2
Water		15		Note 3
Soil K_{oc} , mL/g		180,000		Knissel and Davis 2000
Sediment K_d , mL/g	3870	3030	1250	Note 4
Water Solubility, mg/L		0.005		Tomlin 2005
Foliar wash-off fraction		0.4		Knissel and Davis 2000
Fraction applied to foliage		0.5		Note 5

Note 1 Based on upper bound of 11 hours from Tomlin (2004) rounded upward to 0.5 days.

Note 2 Variable. The 30 day values is taken from Knissel and Davis 2000. Half-times of 23-82 days reported in Tomlin (2004). U.S. EPA/OPP (1988a) reports a half-time of <30 days.

Note 3 Upper bound of 7-15 day values from Tomlin (2004) and USDA/ARS (1995)

Note 4 Based on equation $K_d = K_{oc} \times P_{OC}$ where P_{OC} is the proportion of organic matter in the soil. Estimate P_{OC} as $P_{OM} \times 0.58$ based on P_{OM} values of 0.012 for sand, 0.029 for loam, and 0.037 for clay. All values rounded to 3 significant places

Note 5 The Forest Service estimates that 0.75 may be more appropriate for tree applications but this estimate is not well-documented and 0.5 is used as a more conservative value that is typically used for directed foliar applications.

Table 6: Summary of Gleams-Driver Modeling

Scenario	Concentrations (mg/L) ^a	
	Peak ^c	Long-Term Average ^c
EXPECTED CONCENTRATIONS		
CHICO (COMANCHE CREEK, AIRBLAST) ^b		
0.08 lb a.i./acre, six applications at 2 week intervals, starting in May, including drift	5.35 x 10 ⁻⁶ (4.38 x 10 ⁻⁶ to 2.50 x 10 ⁻⁵) Max: 3.09 x 10 ⁻⁵	1.56 x 10 ⁻⁷ (1.06 x 10 ⁻⁷ to 2.97 x 10 ⁻⁷) Max: 3.40 x 10 ⁻⁷
0.16 lb a.i./acre, three applications at 4 week intervals, starting in May, including drift	1.02 x 10 ⁻⁵ (6.45 x 10 ⁻⁶ to 2.25 x 10 ⁻⁵)	1.60 x 10 ⁻⁷ (9.92 x 10 ⁻⁸ to 2.88 x 10 ⁻⁷)
FORESTHILL (MCBRIDE CREEK, HIGH-PRESSURE SPRAY) ^b		
0.08 lb a.i./acre, six Applications at 2 week intervals, starting in May, including drift	7.73 x 10 ⁻⁶ (1.29 x 10 ⁻⁷ to 5.89 x 10 ⁻⁵) Max: 1.01 x 10 ⁻⁴	8.02 x 10 ⁻⁸ (3.76 x 10 ⁻⁹ to 4.80 x 10 ⁻⁷) Max: 8.00 x 10 ⁻⁷
0.16 lb a.i./acre, three applications at 4 week intervals, starting in May, including drift	6.88 x 10 ⁻⁶ (2.14 x 10 ⁻⁷ to 5.32 x 10 ⁻⁵)	7.28 x 10 ⁻⁸ (3.73 x 10 ⁻⁹ to 4.31 x 10 ⁻⁷)
WATER CONTAMINATION RATES		
SIX APPLICATIONS OF 1 LB A.I./ACRE AT TWO WEEK INTERVALS		
Chico Site (Comanche Creek), 6 applications at 0.08 lb a.i./acre with 2 week interval	6.69 x 10 ⁻⁵ (5.48 x 10 ⁻⁵ to 3.13 x 10 ⁻⁴)	1.95 x 10 ⁻⁶ (1.34 x 10 ⁻⁶ to 3.71 x 10 ⁻⁶)
Chico Site (Comanche Creek), 3 applications at 0.16 lb a.i./acre with 4 week interval	6.40 x 10 ⁻⁵ (4.03 x 10 ⁻⁵ to 1.41 x 10 ⁻⁴)	1.00 x 10 ⁻⁶ (6.20 x 10 ⁻⁷ to 1.80 x 10 ⁻⁶)
Foresthill Site (McBride Creek) 6 applications at 0.08 lb a.i./acre with 2 week interval	9.66 x 10 ⁻⁵ (1.61 x 10 ⁻⁶ to 7.36 x 10 ⁻⁴)	1.00 x 10 ⁻⁶ (4.70 x 10 ⁻⁸ to 6.00 x 10 ⁻⁶)
Foresthill Site (McBride Creek) 3 applications at 0.16 lb a.i./acre with 4 week interval	4.30 x 10 ⁻⁵ (1.34 x 10 ⁻⁶ to 3.33 x 10 ⁻⁴)	4.55 x 10 ⁻⁷ (2.33 x 10 ⁻⁸ to 2.69 x 10 ⁻⁶)
Generic Modeling, Stream (used in WorksheetMaker) ^c	1.75 x 10 ⁻⁴ (1x10 ⁻⁷ . to 7.9 x 10 ⁻³)	4.52 x 10 ⁻⁶ (9.0 x 10 ⁻⁹ to 7.0 x 10 ⁻⁴)
Generic Modeling, Pond ^c	7.0 x 10 ⁻⁴ (1.1x10 ⁻⁷ . to 7.9 x 10 ⁻⁴)	7.0 x 10 ⁻⁷ (1.0x10 ⁻⁹ . to 6.0 x 10 ⁻⁶)

^a Values are given as the median with lower and upper empirical 95% interval followed by the maximum modeled concentrations from 100 simulations.

^b See Section 3.2.3.4.3.2 for discussion.

^c Minimum concentrations are modeled at zero. For this summary table, the lowest non-zero concentration is used.

Table 7: Monitored concentrations of lambda-cyhalothrin in sediment

		Koc Values (kg/L)		
		Clay	Loam	Sand
		3870	3030	1250
Sediment Source	Sediment Concentration (mg/kg)	Estimated concentrations in water (mg/L)		
Pond ^a	0.0168	4.34E-06	5.54E-06	1.34E-05
Irrigation Canals ^a	0.0078	2.02E-06	2.57E-06	6.24E-06
Stream Sediments (Jan) ^b	0.0040	1.03E-06	1.32E-06	3.20E-06
Stream Sediments (Dec) ^b	0.0131	3.39E-06	4.32E-06	1.05E-05

Average Water Concentration: 4.82E-06 Mg/L
 Lower Bound Water Concentration: 1.03E-06 Mg/L
 Upper Bound Water Concentration: 1.34E-05 Mg/L

Source of Sediment Concentrations: ^a Weston et al. 2004; ^b Weston et al. 2008

See Section 3.2.3.4.5 for discussion.

Table 8: Estimated residues in food items per lb a.i. applied

Food Item	Concentration in Food Item (ppm per lb a.i./acre)		
	Central ^a	Lower ^b	Upper ^a
Broadcast Foliar Applications			
Short grass	85	30	240
Tall grass	36	12	110
Broadleaf/forage plants and small insects	45	15	135
Fruits, pods, seeds, and large insects	7	3.2	15
^a From Fletcher et al. (1997) and U.S. EPA/EFED 2001, p. 44.			
^b Calculated as the Central value \times (Central Value \div Upper Value).			

Table 9: Summary of toxicity values used in human health risk assessment			
Duration	Derivation of RfD	Reference	Comment
Acute – single exposure			
NOAEL Dose	0.5 mg/kg bw/day	U.S. EPA/OPP 2003b	This is based on a chronic study in dogs in which ataxia was observed from Day 2 onward at 3 to 7 hours after dosing. The dose of 0.5 mg/kg bw/day is considered a NOAEL for short-term exposures.
LOAEL Dose	3.5 mg/kg bw/day		
LOAEL Endpoint(s)	Ataxia		
Species, sex	Dogs		
Uncertainty Factor	100		
RfD	0.005 mg/kg bw/day		
Chronic – lifetime exposure			
NOAEL Dose	0.1 mg/kg bw/day	U.S. EPA/OPP 2003b	Some signs of neurotoxicity seen at 0.5 mg/kg bw/day in some animals and this dose is classified as a LOAEL in U.S EPA/HED (1997c) but was subsequently reclassified as a NOEL.
LOAEL Dose	0.5 mg/kg bw/day		
Species, sex	Dogs		
LOAEL Endpoint(s)	Abnormal gait		
Uncertainty Factor	100		
RfD	0.001 mg/kg bw/day		

Table 10: Summary of acute toxicity values for fish

Species	96-Hour LC ₅₀ (µg/L)	Reference
<i>Leuciscus idus</i> (golden orfe)	0.078	Maund et al. 1998
<i>Lepomis macrochirus</i> (bluegill sunfish)	0.106	Marino and Rick (2001b)
<i>Ictalurus punctatus</i> (catfish)	0.16	Maund et al. 1998
<i>Oncorhynchus mykiss</i> (rainbow trout)	0.190	Machado 2001a
<i>Lepomis macrochirus</i> (bluegill sunfish)	0.21	U.S. EPA/EFED 1988
<i>Oncorhynchus mykiss</i> (rainbow trout)	0.24	U.S. EPA/EFED 1988
<i>Gasterosteus aculeatus</i> (stickleback)	0.40	Maund et al. 1998
<i>Cyprinus carpio</i> (carp)	0.50	Maund et al. 1998
<i>Brachydanio rerio</i> (zebra fish)	0.64	Maund et al. 1998
<i>Pimephales promelas</i> (fathead minnow)	0.70	Maund et al. 1998
<i>Oryzias latipes</i> (rice fish)	1.4	Maund et al. 1998
<i>Brachydanio rerio</i> (Zebra fish)	1.94	Wang et al. 2007
<i>Poecilia reticulata</i> (guppy)	2.3	Maund et al. 1998
<i>Channa punctatus</i> (channel catfish)	7.92	Kumar et al. 2007

Includes only 96-hour LC₅₀ values for technical grade lambda-cyhalothrin. See Appendix 3 (Table 1) for details and Section 4.1.3.1 for discussion.

Table 11: Summary of acute toxicity values for aquatic invertebrates

Species	48-h EC ₅₀ or LC ₅₀ (µg/L)	Reference
<i>Hyalella azteca</i> (freshwater shrimp)	0.0023	Maund et al. 1998
<i>Chaoborus</i> sp. (phantom midge)	0.0028	Maund et al. 1998
<i>Mysidopsis bahia</i> (mysid shrimp)	0.0041	ECOTOX in Giddings et al. 2009
<i>Gammarus pulex</i>	0.00668	U.S. EPA/OPP 1988a
<i>Gammarus pulex</i> (amphipod)	0.0068	ECOTOX in Giddings et al. 2009
<i>Gammarus pulex</i>	0.00913	U.S. EPA/OPP 1988a
<i>Gammarus pulex</i> (scud)	0.014	Maund et al. 1998
<i>Notonecta glauca</i> (backswimmer)	0.0226	Schroer et al. 2004
<i>Daphnia magna</i> , sensitive population	0.025	Barata et al. 2002
<i>Asellus aquaticus</i> (isopod)	0.026	Maund et al. 1998
<i>Corixa</i> sp. (water boatman)	0.030	Maund et al. 1998
<i>Gammarus pulex</i> (scud)	0.0314	Schroer et al. 2004
<i>Cloeon dipterum</i> (mayfly nymph)	0.038	Maund et al. 1998
<i>Macrobrachium nipponensis</i> (shrimp)	0.04	Wang et al. 2007
<i>Hydracarina</i> (water mite)	0.047	Maund et al. 1998
<i>Sigara striata</i> (Hemiptera)	0.0492	Schroer et al. 2004
<i>Daphnia magna</i>	0.051	Machado 2001b
<i>Proasellus coxalis</i> (isopod)	0.0788	Schroer et al. 2004
<i>Cloeon dipterum</i> (Ephemeroptera)	0.122	Schroer et al. 2004
<i>Ischnura elegans</i> (damselfly)	0.13	Maund et al. 1998
<i>Asellus aquaticus</i> (isopod)	0.140	Schroer et al. 2004
<i>Daphnia magna</i> , tolerant population	0.17	Barata et al. 2002
<i>Caenis horaria</i> (Ephemeroptera)	0.257	Schroer et al. 2004
<i>Cyclops</i> sp. (copepod)	0.30	Maund et al. 1998
<i>Ceriodaphnia dubia</i>	0.3	Mokry and Hoagland 1990
<i>Daphnia magna</i>	0.36	U.S. EPA/OPP 1988a
<i>Daphnia galeata</i> (daphnid)	0.397	Schroer et al. 2004
<i>Macropelopia</i> sp. (chironomid)	1.019	Schroer et al. 2004
<i>Daphnia magna</i>	1.04	Mokry and Hoagland 1990
<i>Simocephalus vetulus</i> (fairly shrimp)	1.340	Schroer et al. 2004
<i>Erythromma viridulum</i> (damselfly)	1.583	Schroer et al. 2004
<i>Chironomus riparius</i> (midge)	2.4	Maund et al. 1998
Ostracoda (<i>seed shrimp</i>) NOS	3.3	Maund et al. 1998
<i>Bithynia tentaculata</i> (snail)	>8.9	Schroer et al. 2004
<i>Polycelis nigra/tenuis</i> (flatworm)	>50	Schroer et al. 2004
<i>Crassostrea gigas</i> (Pacific oyster)	>590	Giddings et al. 2009

Includes only 48-hour LC₅₀ values for technical grade lambda-cyhalothrin. LC₅₀ values in bold indicate that the value was greater than the specified concentration.

Table 12: EC₅₀ Values for Immobilization of *Hyalella azteca* in different pond waters

Water body	Dissolved Organic Carbon (mg/L)	48-h EC₅₀ (µg/L)
UMFS pond S-4	1.0	0.0024
UMFS pond S-2	1.4	0.0017
UMFS pond Bramlett pond	1.7	0.0022
UMFS pond 97	1.9	0.0015
UMFS pond 146	2.7	0.0039
UMFS pond 167	2.8	0.0014
UMFS pond 179	5.3	0.0036
UMFS pond 1	10.6	0.0028
UMFS pond 98	11.4	0.0074
Beasley lake	11.4	0.0111
UMFS pond 92	16.7	0.0104
Coldwater bendway	32.9	0.0157

Data from Smith and Lizotte (2007), Tables 1 and 3.
Data illustrated in Figure 7.
See Section 4.1.3.3 for discussion.

Table 13: Soil modeling for Chico and Foresthill Sites

Parameter	Value
Chico Site	
Peak Soil Concentration, top 12" (from Gleams-Driver)	0.049 (0.048-0.061) mg/kg
Maximum depth of penetration into soil	4 inches
Peak Soil Concentration, top 4"	0.15 (0.14-0.18) mg/kg
Foresthill Site	
Peak Soil Concentration, top 12" (from Gleams-Driver)	0.049 (0.048-0.056) mg/kg
Maximum depth of penetration into soil	4 inches
Peak Soil Concentration, top 4"	0.15 (0.14-0.17) mg/kg

Table 14: Toxicity values used in ecological risk assessment

Group/Duration	Organism	Endpoint	Toxicity Value (a.i.)	Reference
Terrestrial Animals				
Acute				
	Non-canine Mammals	NOEL	10 mg/kg bw	Section 4.3.2.1.
	Canids	NOEL	0.5 mg/kg bw	Section 4.3.2.1.
	Birds	NOEL	150 mg/kg bw	Section 4.3.2.2
	Terrestrial Insects, contact	LD ₅₀ ÷ 10	0.0065 mg/kg bw	Section 4.3.2.3.1
	Terrestrial Insects, oral	LD ₅₀ ÷ 24	0.4 mg/kg bw	Section 4.3.2.3.2
	Earthworm	NOEC	10 to 63.2 ppm soil	Section 4.3.2.3.3
	Soil Nematode	LOEC	0.0002 ppm soil	Section 4.3.2.3.3
	Isopod	LC ₅₀ ÷ 10	0.05 ppm soil	Section 4.3.2.3.3
Longer-term				
	Non-canine Mammals	NOEL	1.5 mg/kg bw/day	Section 4.3.2.1
	Canids	NOEL	0.1 mg/kg bw/day	Section 4.3.2.1
	Bird	NOEL	0.85 mg/kg bw/day	Section 4.3.2.2.
	Earthworm	NOEC	3.2 to 10 ppm soil	Section 4.3.2.3.3
	Soil Microorganisms	NOEC	1 to 20 ppm soil	Section 4.3.2.5
Aquatic Animals				
Acute				
Fish	Sensitive	LC ₅₀ ÷ 20	0.000004 mg/L	Section 4.3.3.1.1
	Tolerant	LC ₅₀ ÷ 20	0.0004 mg/L	Section 4.3.3.1.1
Arthropods	Sensitive	Chronic NOEC	0.0000002 mg/L	Section 4.3.3.3.1
	Tolerant	Chronic NOEC	0.00017 mg/L	Section 4.3.3.3.1
Other Invertebrates	Sensitive	LOEC ÷ 10	0.0009 mg/L	Section 4.3.3.3.2
	Tolerant	>LC ₅₀ ÷ 10	0.06 mg/L	Section 4.3.3.3.2
Longer-term				
Fish	Sensitive	Estimated life-cycle NOEC	0.0000031 mg/L	Section 4.3.3.1.2
	Tolerant	Estimated life-cycle NOEC	0.00031 mg/L	Section 4.3.3.1.2
Arthropods	Sensitive	NOEC	0.0000002 mg/L	Section 4.3.3.3.1
	Tolerant	LC ₅₀ ÷ 20	0.00017 mg/L	Section 4.3.3.3.1
Other Invertebrates	Sensitive	No data	N/A	Section 4.3.3.3.2
	Tolerant	No data	N/A	Section 4.3.3.3.2

Appendix 1: Information on mammalian toxicity from MSDSs (continued)

Appendix 1: Information on mammalian toxicity from MSDSs

A1 Table 1: Oral LD₅₀ data from Material Safety Data Sheets

Formulation	a.i	LD ₅₀ (mg/kg bw)	
		MSDS	a.i.
Grizzly Z	11.4%	351	40
Kaiso 24 WG	24%	310	74.4
Lambda-Cy EC	11.4%	81	9.234
LambdaStar 1 CS	12.0%	60 -190	7.2-22.8
Lambda T	11.4%	351	40
Silencer	12.7%	98.11	12.46
Taiga Z	11.4%	351	40
Warrior with Zeon Technology	11.4%	351	40
Warrior II with Zeon Technology	22.8%	180*	41.04

*Based on a similar product.

U.S. EPA reports oral LD₅₀ values of 56 mg/kg (♀) and 79 mg/kg (♂) for technical grade lambda-cyhalothrin (HED 2002).

A1 Table 2: Dermal LD₅₀ data from Material Safety Data Sheets

Formulation	a.i	LD ₅₀ (mg/kg bw)	
		MSDS	a.i.
Grizzly Z	11.4%	>2,000	>288
Kaiso 24 WG	24%	>5,000	>1,200
Lambda-Cy EC	11.4%	>2,000	>288
LambdaStar 1 CS	12.0%	>2,000	>240
Lambda T	11.4%	>2,000	>288
Silencer	12.7%	>2,000	>254
Taiga Z	11.4%	>2,000	>288
Warrior with Zeon Technology	11.4%	>2,000	>228
Warrior II with Zeon Technology	22.8%	>2,000*	>456

* Based on a similar product.

U.S. EPA reports dermal LD₅₀ values of 696 mg/kg (♀) and 632 mg/kg (♂) for technical grade lambda-cyhalothrin (HED 2002).

Appendix 1: Information on mammalian toxicity from MSDSs (continued)

A1 Table 3: Inhalation 4-hour LC₅₀ data from Material Safety Data Sheets

Formulation	a.i	LC ₅₀ (mg/L)	
		MSDS	a.i.
Grizzly Z	11.4%	2.5	0.285
Kaiso 24 WG	24%	N/A	N/A
Lambda-Cy EC	11.4%	0.622	0.071
LambdaStar 1 CS	12.0%	>2.2	>0.264
Lambda T	11.4%	>2.5	>0.285
Silencer	12.7%	1.83	0.232
Taiga Z	11.4%	2.5	0.285
Warrior with Zeon Technology	11.4%	>2.5	>0.285
Warrior II with Zeon Technology	22.8%	3.12	0.711

U.S. EPA reports inhalation LD₅₀ values of 0.065 mg/L (♀ and ♂) for technical grade lambda-cyhalothrin (HED 2002).

A1 Table 4: Eye contact information

Formulation	a.i	Information form MSDS
Grizzly Z	11.4%	May cause mild eye irritation. Mild eye irritation in rabbits.
Kaiso 24 WG	24%	Substantial but temporary eye irritation. Severe eye irritation in rabbits.
Lambda-Cy EC	11.4%	Mild eye irritation in rabbits.
LambdaStar 1 CS	12.0%	Mild eye irritation in rabbits.
Lambda T	11.4%	Slight eye irritation.
Silencer	12.7%	Substantial but temporary eye irritation. Mildly irritating in eye irritation studies using rabbits.
Taiga Z	11.4%	Mild eye irritation in rabbits.
Warrior with Zeon Technology	11.4%	Mild eye irritation.
Warrior II with Zeon Technology	22.8%	Mild eye irritation in rabbits.

Appendix 1: Information on mammalian toxicity from MSDSs (continued)

A1 Table 5: Skin contact information

Formulation	a.i	Information form MSDS
Grizzly Z	11.4%	May cause mild skin irritation. Toxic if absorbed through skin. Slightly irritating in rabbits.
Kaiso 24 WG	24%	Moderate skin irritation. May cause a temporary itching, tingling, burning or numbness of exposed skin, called paresthesia. Skin contact may aggravate existing skin disease.
Lambda-Cy EC	11.4%	Slight skin irritation in rabbits.
LambdaStar 1 CS	12.0%	Mildly irritation in rabbit dermal irritation studies. Irritation in humans can result after repeated and/or prolonged contact. May result in tingling, itching, burning, or prickly feeling.
Lambda T	11.4%	Slight skin irritation in rabbits. Prolonged contact may cause mild to moderate irritation.
Silencer	12.7%	Slight skin irritation in rabbits. Skin exposure may result in a sensation described as a tingling, itching, burning, or prickly feeling.
Taiga Z	11.4%	Slight skin irritation in rabbits. May cause temporary itching, tingling, burning or numbness of exposed skin, called paresthesia.
Warrior with Zeon Technology	11.4%	Slight skin irritation in rabbits. In humans, may cause temporary itching, tingling, burning or numbness, called paresthesia. Face and genital areas are especially susceptible to this effect.
Warrior II with Zeon Technology	22.8%	A similar product causes mild skin irritation in rabbits.

Appendix 1: Information on mammalian toxicity from MSDSs (continued)

A1 Table 6: Skin sensitization information

Formulation	a.i	Information form MSDS
Grizzly Z	11.4%	No information available.
Kaiso 24 WG	24%	Skin sensitization in guinea pigs. Frequently repeated skin contact may cause allergic reaction in some individuals.
Lambda-Cy EC	11.4%	No information given on MSDS.
LambdaStar 1 CS	12.0%	No information given on MSDS.
Lambda T	11.4%	This material was a skin sensitizer in animal testing.
Silencer	12.7%	Positive in assay for dermal sensitization. Prolonged or frequently repeated skin contact may cause allergic reaction in some individuals.
Taiga Z	11.4%	No information available. Prolonged skin contact may cause allergic reactions.
Warrior with Zeon Technology	11.4%	No information available.
Warrior II with Zeon Technology	22.8%	A skin sensitizer (derived from components). May cause allergic skin reactions.

Appendix 2: Toxicity to Terrestrial Invertebrates (continued)

Appendix 2: Toxicity to terrestrial invertebrates

A2 Table 1: Acute and Chronic Toxicity Assays 145
 A2 Table 2: Field Studies 148

A2 Table 1: Acute and Chronic Toxicity Assays																								
Species	Exposure	Response	Reference																					
Direct Contact/Topical Application Assays																								
<i>Erigone atra</i> (spider)	Direct contact assay	LD ₅₀ M: 0.33 µg/kg bw F: 0.31 µg/kg bw	Dinter and Poehling 1995																					
<i>Oedothorax apicatus</i> (spider)	Direct contact assay	LD ₅₀ M: 0.44 µg/kg bw F: 0.45 µg/kg bw	Dinter and Poehling 1995																					
<i>Apis mellifera</i> (honey bees) BW not specified, assume 0.000093 kg	Direct contact bioassay. Diethyl maleate and S,S,S-tributylphosphorotrithioate were also tested. Less effective than PBO.	LD ₅₀ alone: 102 (73.0 – 133) ng/bee [0.102 µg/bee or about 1.1 mg/kg bw] LD ₅₀ with piperonyl butoxide: 1.28 (1.12 – 1.46) ng/bee [0.00128 µg/bee]	Johnson et al. 2006																					
<i>Apis mellifera</i> (honey bees), average bw: 126.5 mg	Direct contact assay	LD ₅₀ = 0.022 µg/bee ≈1.74 mg/kg bw	Mayer et al. 1998																					
<i>Nomia melanderi</i> (alkali bee), average bw: 86.6 mg	Direct contact assay	LD ₅₀ = 0.036 µg/bee ≈0.41 mg/kg bw	Mayer et al. 1998																					
<i>Megachile rotundata</i> (alfalfa leafcutter bee), average bw: 30.8 mg	Direct contact assay	LD ₅₀ = 0.002 µg/bee ≈0.065 mg/kg bw	Mayer et al. 1998																					
<i>Apis mellifera</i> (honey bees) BW not specified, assume 0.000093 kg (USDA/APHIS 1993)	Direct contact assay	LD ₅₀ = 68 ng/bee (0.068 µg/bee) ≈0.731 mg/kg bw Toxicity to bees synergized by fungicides.	Pilling and Jepson 1993																					
<i>Apis mellifera</i> (honey bees) BW not specified, assume 0.000093 kg	Direct contact assay. 0.2, 0.1, 0.05, 0.02, 0.01, and 0.005 µg a.i. per bee. Apparently no untreated control group. Make conservative assumption of 0/60 for no treatment in Fisher Exact Test.	48-h LD ₅₀ = 0.038 µg/bee ≈0.409 mg/kg bw Dose/response as 48-hours <table border="1" style="margin-left: auto; margin-right: auto;"> <thead> <tr> <th>Dose</th> <th>N</th> <th>R</th> </tr> </thead> <tbody> <tr> <td>0.005</td> <td>59</td> <td>2</td> </tr> <tr> <td>0.01</td> <td>59</td> <td>2</td> </tr> <tr> <td>0.02</td> <td>59</td> <td>10</td> </tr> <tr> <td>0.05</td> <td>59</td> <td>42</td> </tr> <tr> <td>0.1</td> <td>59</td> <td>48</td> </tr> <tr> <td>0.2</td> <td>59</td> <td>58</td> </tr> </tbody> </table> N: number tested; R: number dead. Fisher Exact: 0/59 vs 2/59 = 0.247863 NOEC for mortality: 0.01 µg/bee. ≈ 0.11 mg/kg bw.	Dose	N	R	0.005	59	2	0.01	59	2	0.02	59	10	0.05	59	42	0.1	59	48	0.2	59	58	Gough et al. 1984 Used by U.S. EPA/EFED 1988
Dose	N	R																						
0.005	59	2																						
0.01	59	2																						
0.02	59	10																						
0.05	59	42																						
0.1	59	48																						
0.2	59	58																						

Appendix 2: Toxicity to Terrestrial Invertebrates (continued)

Residue Contact Assays			
<i>Orius insidiosus</i> (predatory flower bug)	Vegetation contact. 28 g/ha (≈0.025 lb a.i./ac) on corn. . Residue exposures for 3 days.	Significant increase in mortality (See Table 1 in study).	Al-Deeb et al. 2001
<i>Orius insidiosus</i> (predatory flower bug)	Vegetation contact. 560 g/ha (≈0.5 lb a.i./ac) on sorghum. Residue exposures for 3 days	Significant increase in mortality (See Table 7 in study).	Al-Deeb et al. 2001
<i>Micromus tasmaniae</i> (lacewings, predator species) and <i>Rhopalosiphon padi</i> (bird cherry-oat aphid, target species)	Contact with filter paper in petri dishes treated with lambda-cyhalothrin at concentrations of 0.1 to 10 mg/L.	LC ₅₀ s Lacewings: >10 mg/L Aphids: ≈1.1 mg/L	Booth et al. 2007
<i>Hyaliodes vitripennis</i> (mite predator)	Indirect contact assays after insects and vegetation sprayed to runoff with varying concentrations. Data taken from Table 1 of paper	Nymphs LC ₅₀ : 2.3 (1.8-2.9) mg a.i./L Adults LC ₅₀ : 0.7 (0.5-0.9) mg a.i./L	Bostanian et al. 2001
<i>Aphidius ervi</i> (parasitic wasp)	Indirect contact assay, glass	LD ₅₀ : 4.97 ng/cm ² Corresponds to 0.049 μg/cm ² or about 0.0044 lb a.i./acre.	Desneux et al. 2004b
<i>Aphis mellifera</i> (honeybees), 50 per assay	015 and 35 g a.i./ha (0.013 and 0.031 lb a.i./ac) to alfalfa. Foliage collected at 3, 8, 24, 48, and 96 hours. Bees exposed to residues for 24 hours with uncontaminated sucrose.	LT ₅₀ : 23 hours at lower application rate. LT ₅₀ : 4 to 12 hours at higher application rate	Gough and Brown 1987
<i>Aphidius rhopalosiphi</i> (parasitic hymenoptera)	Glass plate contact assay	5 g/ha: 100% mortality. Corresponds to 0.0045 lb/ac.	Jansen 1996
<i>Anystis baccarum</i> (predatory mite)	Indirect Contact LC ₅₀ .	LC ₅₀ : 0.7 mg/L Working note: Cannot determine application in mass/surface area.	Laurin and Bostanian 2007
<i>Apis mellifera</i> (honey bees)	Contact with foliage treated at 0.011 to 0.034 kg/ha that was collected 2 and 8 hours after treatment.	Approximate LD ₅₀ : between 0.028 and 0.034 kg/ha at 2 and 8 h. LD ₅₀ s correspond to 0.024 and 0.03 lb/ac.	Mayer et al. 1998
<i>Nomia melanderi</i> (alkali bee)	Contact with foliage treated at 0.011 to 0.034 kg/ha that was collected 2 and 8 hours after treatment.	Approximate LD ₅₀ : 0.028 kg/ha at 2 hr and between 0.028 and 0.034 kg/ha at 8 h.	Mayer et al. 1998
<i>Megachile rotundata</i> (alfalfa leafcutter bee)	Contact with foliage treated at 0.011 to 0.034 kg/ha that was collected 2 and 8 hours after treatment.	Approximate LC ₅₀ : between 0.022 and 0.028 kg/ha at 2 and 8 h.	Mayer et al. 1998

Appendix 2: Toxicity to Terrestrial Invertebrates (continued)

Dietary Assays			
<i>Spodoptera litura</i> (tobacco cutworm, target species)	Dietary feeding: Cauliflower leaves dipped in solutions of 0, 5, 7.5, 10, 12.5 and 25 ppm and then fed to insects. Concentrations on leaves not specified. Cannot approximated mg/kg bw dose.	NOAEL: Not identified LOAEL: 5 ppm for survival and several other endpoints. Note: the concentrations refer to liquid used to treat leaves and not to resulting concentration on leaves.	Abro et al. 1997
<i>Apis mellifera</i> (honey bees), assume 0.000093 kg (USDA/APHIS 1993)	Oral LD ₅₀ with contaminated sucrose solution. Doses: 0.0025, 0.005, 0.01, 0.025, 0.05, 0.1 and 0.25 µg/bee.	48-h LD ₅₀ = 0.909 µg/bee ≈9.774 mg/kg bw Note: The DER does not give the dose-response data for the oral assay.	Gough et al. 1984 Used by U.S. EPA/EFED 1988
Soil Assays			
<i>Eisenia fetida</i> (earthworm)	Avoidance behavior in tropical artificial soil, ≈6% OM (TAS)	Acute LC ₅₀ /NOEC: 23.9 ppm/10 ppm Chronic (repro) EC ₅₀ /NOEC: 4.6 ppm/3.2 ppm Avoidance EC ₅₀ /NOEC: 0.2 ppm/<0.3 ppm	Garcia et al. 2008
<i>Eisenia fetida</i> (earthworm)	Avoidance behavior in ≈6.17% OM peat moss (OECD soil)	Acute LC ₅₀ /NOEC: 99.8 ppm/63.2 ppm Chronic (repro) EC ₅₀ /NOEC: 37.4 ppm/10 ppm Avoidance EC ₅₀ /NOEC: 3.3 ppm/1 ppm	Garcia et al. 2008
<i>Eisenia fetida</i> (earthworm)	Avoidance behavior in European natural field soil, ≈4.6% OM (LUFÄ)	Acute LC ₅₀ /NOEC: 139.9 ppm/31.6 ppm Chronic (repro) EC ₅₀ /NOEC: 44.5 ppm/3.216 ppm Avoidance EC ₅₀ /NOEC: 0.5 ppm/<0.3 ppm	Garcia et al. 2008
<i>Caenorhabditis elegans</i> (soil nematode)	Soil concentrations of 0.002, 0.02, 0.2, or 2 mg/L.	Concentration dependant decrease in locomotion. No apparent NOEC. Very slight decrease in brood size. Effect on body length only at highest concentration. No statistics in this paper.	Ruan et al. 2009
<i>Eisenia fetida</i> (earthworm)	Soil LC ₅₀	265.5 mg/kg soil	Frampton et al. 2006
<i>Porcellionides pruinosus</i> (woodlouse, Isopoda)	Soil LC ₅₀	≈ 0.5 mg/kg soil (most sensitive species; read from Figure 2, p. 2485)	Frampton et al. 2006

Appendix 2: Toxicity to Terrestrial Invertebrates (*continued*)

A2 Table 2: Field/Mesocosm Studies			
Species	Exposure	Response	Reference
<i>Apis mellifera</i> (honeybees)	Mesocosm: 7.5 or 15 g a.i./ha (0.0067 or 0.013 lb a.i./ac) to winter wheat sprayed with sucrose to simulate foraging. Bees had access to treated or untreated media.	Inhibition of foraging for 3 days with marked inhibition on Day 1. Toxicity cannot be assessed directly. This is an avoidance study.	Gough et al. 1986
<i>Apis mellifera</i> (honeybees)	0.0075 and 0.015 lb a.i./ac to seed alfalfa.	Mortality on direct exposure: ≈50% at lower rate and ≈90% at higher rate. Decreases (≈40 to 50%) in <i>bee visitation</i> for two days after application. The DER does not include a good description of other study observations – i.e., colony strength.	Hearn 1985
<i>Tetranychus urticae</i> (two spotted spider mite)	Application rates: 2.5 g a.i./ha [≈0.002 lb/ac] and 6.25 g a.i./ha [≈0.005 lb/ac] in 1988 and 1990, and 1.25 g a.i./ha [≈0.001 lb/ac] and 2.5g a.i./ha [≈0.002 lb/ac] in 1989.	Significant increase in mite populations. Possibly due to repellent properties of lambda-cyhalothrin, resulting in an increase dispersion of mites.	Li and Harmsen 1993
<i>Megachile rotundata</i> (alfalfa leafcutter bee)	Application rates: To alfalfa at 0.011, 0.17, 0.28 kg a.i./ha. Equivalent to approximately 0.001, 0.15, and 0.25 lb a.i./ac.	Reduced bee populations only at highest application rate.	Mayer et al. 1998
Various species of spiders and rove beetles (<i>Tachyporus hypnorum</i>)	Application rates: Treatment of winter wheat at 2.5, 5, and 10 g/ha [≈ 0.0022, 0.0045, and 0.0089 lb a.i./acre]	Substantial changes in species composition at highest application rate. Decrease in abundance of rove beetles. Increase in abundance of some groups of spiders but a decrease in abundance of other spider species. Impact on larval development of rove beetle) at lowest application rate.	Niehoff et al. 1994

Appendix 3: Toxicity to fish

A3 Table 1: Acute Toxicity 149
 A3 Table 2: Chronic toxicity 150
 A3 Table 3: Field/Mesocosm Studies 151

A3 Table 1: Acute Toxicity				
Species	Exposure	Response		Reference
<i>Cyprinodon variegatus</i> (sheepshead minnow)	96-hour, flow-through. Measured concentrations of 0.29, 0.55, 1.35, 1.72, and 2.37 µg/L.	Hours	LC ₅₀ in µg/L	Hill 1985d
		24	1.34	
		48	1.14	
		72	0.85	
		96	0.81	
		NOEC for mortality and signs of toxicity = 0.29 µg/L. 96-hLC ₅₀ /NOEC= 2.7		
<i>Brachydanio rerio</i> (Zebra fish)	96 hours, different formulation specific LC ₅₀ s. Not clear that toxicity values are expressed in formulation or a.i.	Formulation	96-h LC ₅₀ µg/L	Gu et al. 2007
		2.5% ME	1.21	
		2.5% EW	1.28	
		2.5% SC	7.55	
		2.9% EC	0.98	
		5% WP	1.30	
<i>Channa punctatus</i> (channel catfish)	96-h, static with 24 h renewal, 5% EC formulation. 2.5, 5, 7.5, 10, 12.5 and 15 µg/L.	LC ₅₀ : 7.92 µg/L At 2.5 µg/L, 1/12 died vs 0/12 in the control group (p=0.5). In a separate assay, signs of toxicity including hyperactivity and increased opercular activity observed at 0.8 µg/L (Table 5 of study).		Kumar et al. 2007
<i>Oncorhynchus mykiss</i> (rainbow trout)	96-h	LC ₅₀ : 0.190 µg/L		Machado 2001a
<i>Lepomis macrochirus</i> (bluegill sunfish)	96-h	LC ₅₀ : 0.106 µg/L		Marino and Rick (2001b)
<i>Gasterosteus aculeatus</i> (stickleback)	96-h	LC ₅₀ : 0.40 µg/L		Maund et al. 1998
<i>Oryzias latipes</i> (rice fish)	96-h	LC ₅₀ : 1.4 µg/L		Maund et al. 1998
<i>Ictalurus punctatus</i> (catfish)	96-h	LC ₅₀ : 0.16 µg/L		Maund et al. 1998
<i>Brachydanio rerio</i> (zebra fish)	96-h	LC ₅₀ : 0.64 µg/L		Maund et al. 1998
<i>Poecilia reticulata</i> (guppy)	96-h	LC ₅₀ : 2.3 µg/L		Maund et al. 1998
<i>Leuciscus idus</i> (golden orfe)	96-h	LC ₅₀ : 0.078 µg/L		Maund et al. 1998

Appendix 3: Toxicity to fish (*continued*)

A3 Table 1: Acute Toxicity				
Species	Exposure	Response		Reference
<i>Pimephales promelas</i> (fathead minnow)	96-h	LC ₅₀ : 0.70 µg/L		Maund et al. 1998
<i>Cyprinus carpio</i> (carp)	96-h Separate study on sediment binding and toxicity. See Section 4.1.3.1 for discussion.	LC ₅₀ : 0.50 µg/L		Maund et al. 1998
<i>Brachydanio rerio</i> (Zebra fish)	Static with 24-h renewal.	Hours	LC ₅₀ in µg/L	Wang et al. 2007
		24	8.26	
		48	3.91	
		72	2.05	
		96	1.94	

A3 Table 2: Chronic toxicity			
Species	Exposure	Response	Reference
<i>Cyprinodon variegatus</i> (sheepshead minnow)	Egg-to-fry study Observations up to 32 days post-hatch.	NOEC: 0.25 µg/L LOEC: 0.38 µg/L based on decreased body weight. This is a registrant submitted study and is classified as Core by U.S. EPA/EFED (1998). The 1987 DER for this study, however, classifies the study as supplemental.	Hill et al. 1985
<i>Pimephales promelas</i> (Fathead minnow)	Full life-cycle study. Nominal Concentrations of 0, 0.03, 0.06, 0.12, 0.25, and 0.5 µg/L. Design: F0: 60 day pre-spawning to 300 days spawning. F1: 56 days post-hatch.	NOECs from DER: F0 Embryos Hatching: 0.273 µg/L F0 Larval Survival by Days 28 and 56: 0.062 µg/L F0 Day 300 Weight: NOEC: 0.031 µg/L (measured) LOEC: 0.062 µg/L (measured) based on growth/egg production. F1: Embryo/Larval Survival NOEC: 0.031 µg/L LOEC: 0.062 µg/L See Section 4.1.3.1.2 for discussion.	Tapp et al. 1990

Appendix 3: Toxicity to fish (*continued*)

A3 Table 3: Field/Mesocosm Studies			
Species	Exposure	Response	Reference
<i>Tilapia nilotica</i> (Nile tilapia) in rice paddies in the Philippines.	Low rate: 6.25 g a.i./ha at 15 and 30 days after planting followed by 12.5 g a.i./ha at 45, 60, and 75 days after planting. High rate: 12.5 g a.i./ha at 15 and 30 days after planting followed by 25 g a.i./ha at 45, 60, and 75 days after planting.	Monitored water concentrations not given. At 1.5 g a.i./ha, the nominal concentration is specified as 12.5 µg/L. No adverse effects on fish recovery or growth. No treatment-related effects on fish production.	Hamer et al. 1994
U.S. Mesocosms: 16 450 m ³ mesocosms. Mixed populations of pond-collected fish (bluegills with some minnows), macroinvertebrates, and macrophytes.	Lambda-cyhalothrin applied in multiple times. High rate: Peak concentrations of about 0.09 to 0.1 µg/L. Mid rate: Peak concentration of about 0.01 µg/L. Low rate: Below the limit of detection (0.001 µg/L). 7 month period observation.	Treated mesocosms contained greater numbers of fish (17-20%) than control mesocosms. The biomass of young fish was significantly less (28-38%) than in control mesocosms.	Hill et al. 1994 (also summarized in Hamer et al. 1994)
<i>Gambusia affinis</i> (mosquitofish) in rice paddies in California.	Application rate of 5.8 g/ha (0.58 mg/m ²). Depth of water not precisely specified but it appears that the water was about 20 cm deep (p. 431, column 2, first full paragraph). Observation period of less than 1 month.	Estimated concentration in water is 7.92 µg/L. See Section 4.1.3.1.3 for discussion. Substantial decrease in numbers of living mosquitofish. See Figure 1 of paper.	Lawler et al. 2003

Appendix 4: Toxicity to aquatic invertebrates.

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A4 Table 1: Acute toxicity																	
Species	Exposure	Response	Reference														
<i>Hyalella azteca</i> (amphipod)	American River water, 10 day sediment toxicity bioassay	Sediment LC ₅₀ : 6 ng/g dry sediment weight Sediment LC ₅₀ : 0.14 µg/g OC dry weight LOEC: 0.14 µg/g OC, 35% reduction in biomass	Amweg et al. 2005														
<i>Hyalella azteca</i> (amphipod)	Del Puerto Creek, 10 day sediment toxicity bioassay. Working Note: Concentrations given as sediment or OC and not water column.	Sediment LC ₅₀ : 5.2 ng/g dry sediment weight Sediment LC ₅₀ : 0.46 µg/g OC dry weight. LOEC: 0.23 µg/g OC, 36% reduction in biomass	Amweg et al. 2005														
<i>Daphnia magna</i>	lambda-cyhalothrin, technical grade	Determined EC ₅₀ and EC ₁₀ values in clones of three different populations. See Figure 2 of paper. EC _{10S} : ≈ 0.01 – 0.07 µg/L EC _{50S} : ≈ 0.025 – 0.17 µg/L	Barata et al. 2002														
<i>Daphnia magna</i>	Technical grade cyhalothrin. Not clear that this is the lambda-cyhalothrin mixture. Assayed for effects on dissolved organic carbon (DOC).	<table border="1"> <thead> <tr> <th rowspan="2">DOC mg/L</th> <th colspan="2">LC₅₀ (µg/L)</th> </tr> <tr> <th>24-h</th> <th>48-h</th> </tr> </thead> <tbody> <tr> <td>1.3</td> <td>0.61</td> <td>0.19</td> </tr> <tr> <td>6.7</td> <td>2.91</td> <td>0.18</td> </tr> <tr> <td>9.7</td> <td>2.86</td> <td>0.33</td> </tr> </tbody> </table> <p>See study by Smith and Lizotte 2007 for similar pattern.</p>	DOC mg/L	LC ₅₀ (µg/L)		24-h	48-h	1.3	0.61	0.19	6.7	2.91	0.18	9.7	2.86	0.33	Day 1991
DOC mg/L	LC ₅₀ (µg/L)																
	24-h	48-h															
1.3	0.61	0.19															
6.7	2.91	0.18															
9.7	2.86	0.33															
<i>Gammarus pulex</i> (amphipod)	48-h	EC ₅₀ : 6.8 ng/L (0.0068 µg/L)	Giddings et al. 2009														
<i>Mysidopsis bahia</i> (mysid shrimp)	48-h	EC ₅₀ : 4.1 ng/L (0.0041 µg/L)	Giddings et al. 2009														
<i>Crassostrea gigas</i> (Pacific oyster)	48-h	EC ₅₀ : >590 µg/L	Giddings et al. 2009														
<i>Macrobrachium nipponensis</i> (shrimp)	96 hours, different formulation specific LC _{50S} . Not clear that toxicity values are expressed in formulation or a.i.	<table border="1"> <thead> <tr> <th>Formulation</th> <th>96-h LC₅₀ µg/L</th> </tr> </thead> <tbody> <tr> <td>2.5% ME</td> <td>1.21</td> </tr> <tr> <td>2.5% EW</td> <td>1.28</td> </tr> <tr> <td>2.5% SC</td> <td>7.55</td> </tr> <tr> <td>2.9% EC</td> <td>0.98</td> </tr> <tr> <td>5% WP</td> <td>1.30</td> </tr> </tbody> </table>	Formulation	96-h LC ₅₀ µg/L	2.5% ME	1.21	2.5% EW	1.28	2.5% SC	7.55	2.9% EC	0.98	5% WP	1.30	Gu et al. 2007		
Formulation	96-h LC ₅₀ µg/L																
2.5% ME	1.21																
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2.9% EC	0.98																
5% WP	1.30																

Appendix 4: Toxicity to Aquatic Invertebrates (continued)

<i>Gammarus pulex</i> (amphipod)	EC formulation of lambda-cyhalothrin. Exposure periods of only 0.5 hours – i.e., pulse exposures.	LC ₅₀ : 5.69 (5.14-6.25) µg/L NOEC (mortality): 0.05 µg/L Note: The NOEC is not from the LC ₅₀ study. The lowest dose in the LC ₅₀ study was 0.4 µg/L at which 13.3% mortality was observed. The 0.05 µg/L NOEC for mortality is from separate assay involving a 30 minute pulse exposure (Fig 2 of paper). The NOEC of 0.05 µg/L is based on a very small sample size (n=9). Effects on pre-copulatory behavior at 0.04 µg/L	Heckmann et al. 2005	
<i>Daphnia magna</i>	48-h static renewal	EC ₅₀ : 51 ng/L (0.051 µg/L)	Machado 2001b	
<i>Daphnia magna</i>	72-h in water column or water column/sediment systems	System	72-h LC ₅₀	Maund et al. 1998
		Water	0.26 µg/L	
		Sediment	31.0 µg/L	
		Soil	19.0 µg/L	
<i>Hyalella azteca</i> (freshwater shrimp)	48-h	EC ₅₀ : 0.0023 µg/L	Maund et al. 1998	
<i>Chaoborus</i> sp. (phantom midge)	48-h	EC ₅₀ : 0.0028 µg/L	Maund et al. 1998	
<i>Gammarus pulex</i> (scud)	48-h	EC ₅₀ : 0.014 µg/L	Maund et al. 1998	
<i>Asellus aquaticus</i> (isopod)	48-h	EC ₅₀ : 0.026 µg/L	Maund et al. 1998	
<i>Corixa</i> sp. (water boatman)	48-h	EC ₅₀ : 0.030 µg/L	Maund et al. 1998	
<i>Cloeon dipterum</i> (mayfly nymph)	48-h	EC ₅₀ : 0.038 µg/L	Maund et al. 1998	
Hydracarina (water mite)	48-h	EC ₅₀ : 0.047 µg/L	Maund et al. 1998	
<i>Ischnura elegans</i> (damselfly)	48-h	EC ₅₀ : 0.13 µg/L	Maund et al. 1998	
<i>Cyclops</i> sp. (copepod)	48-h	EC ₅₀ : 0.30 µg/L	Maund et al. 1998	
<i>Chironomus riparius</i> (midge)	48-h	EC ₅₀ : 2.4 µg/L	Maund et al. 1998	
Ostracoda (<i>seed shrimp</i>)	48-h	EC ₅₀ : 3.3 µg/L	Maund et al. 1998	
<i>Daphnia magna</i>	48 h static	LC ₅₀ : 1.04 (0.52-2.94) µg/L	Mokry and Hoagland 1990	
<i>Ceriodaphnia dubia</i>	48 h static	LC ₅₀ : 0.3 (0.15-0.55) µg/L	Mokry and Hoagland 1990	

Appendix 4: Toxicity to Aquatic Invertebrates (continued)

<i>Chaoborus obscuripes</i> (phantom midge)	96-h static Note: units are ng not µg.	Duration	EC50 ng/L	LC50 ng/L	Schroer et al. 2004
		48-h	2.8	>27.4	
		96-h	2.8	75.7	
<i>Notonecta glauca</i> (backswimmer)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	14.8	22.6	
		96-h	16.4		
<i>Proasellus coxalis</i> (isopod)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	17.7	78.8	
		96-h	27.4	44.6	
<i>Caenis horaria</i> (Ephemeroptera)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	17.9	257	
		96-h	13.6	34.6	
<i>Sigara striata</i> (Hemiptera)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	18.2	49.2	
<i>Gammarus pulex</i> (scud)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	23.6	31.4	
		96-h	24.2	24.2	
<i>Asellus aquaticus</i> (isopod)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	24.8	140	
		96-h	24.8	75.2	
<i>Cloeon dipterum</i> (Ephemeroptera)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	24.8	122	
		96-h	88.3	105	
<i>Sialis lutaria</i> (Alderfly)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	51.5	>2179	
		96-h	28	>2179	
<i>Daphnia galeata</i> (daphnid)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	117	397	
<i>Macropelopia</i> sp. (chironomid)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	244	1019	
		96-h	63.4	698	
<i>Erythromma viridulum</i> (damselfly)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	689	1583	
		96-h	493	493	
<i>Simocephalus vetulus</i> (fairy shrimp)	96-h static Note: units are ng not µg.	Duration	EC ₅₀ ng/L	LC ₅₀ ng/L	Schroer et al. 2004
		48-h	957	1340	
<i>Bithynia tentaculata</i> (snail)	96-h static	8900 ng/L. Organisms closed operculum.			Schroer et al. 2004
<i>Lymnaea stagnalis</i> (snail)	96-h static	No concentration-response relationship over the range of 207 to 20,129 ng/L.			Schroer et al. 2004
<i>Polycelis nigra/tenuis</i> (flatworm)	96-h static	No concentration-response relationship over the range of 226 to 30,759 ng/L.			Schroer et al. 2004
<i>Hyaella azteca</i> (amphipod)	48 h static bioassays in waters with difference concentrations of organic matter.	48 h-EC ₅₀ values ranging from 1.7 ng/L to 15.7 ng/L. Significant inverse correlation of toxicity with organic carbon as well as chlorophyll a, turbidity, and suspended solids. See Figure 7 in the current Forest			Smith and Lizotte 2007

Appendix 4: Toxicity to Aquatic Invertebrates (continued)

		Service risk assessment.		
<i>Caridina laevis</i> (freshwater shrimp)	Static exposures	24-h LC ₅₀ : 0.87 (0.76-0.98) µg/L 96-h LC ₅₀ : 0.33 (0.30-0.37) µg/L 96-h NOEC: 0.1 µg/L 96-h LOEC: 0.2 µg/L		Sucahyo et al. 2008
<i>Daphnia magna</i>	N.S.	LC ₅₀ : 0.36 µg/L		U.S. EPA/OPP 1988a
<i>Gammarus pulex</i>	N.S.	2 LC ₅₀ values reported 6.68 ng/L (0.00668 µg/L) 9.13 ng/L (0.00913 µg/L)		U.S. EPA/OPP 1988a
<i>Macrobrachium nippoensis</i> (shrimp)	Static with 24-h renewal.	Hours	LC ₅₀ in µg/L	Wang et al. 2007
		24	0.05	
		48	0.04	
		72	0.04	
		96	0.04	

Appendix 4: Toxicity to Aquatic Invertebrates (*continued*)

A4 Table 2: Chronic toxicity			
Species	Exposure	Response	Reference
<i>Daphnia magna</i>	Technical grade, 0.1 to 0.5 nmol/L. 10-day reproduction study. (Based on a MW of 449.9, concentrations were about 0.045 µg/L to 0.225 µg/L).	Feeding: no NOEC. LOEC 0.1 nmol/L (or 0.045 µg/L). Reproduction: Apparent NOEC of 0.045 µg/L with LOEC of about 0.135 µg/L. See Fig. 1 in paper.	Barata et al. 2006 and Barata et al. 2007
<i>Daphnia magna</i>	Details not specified but presumably a standard 21-day study.	Reproduction: NOEC: 8.5 ng/L (0.0085 µg/L) LOEC: 18.3 ng/L (0.0183 µg/L) Growth: NOEC: 18.3 ng/L (0.0183 µg/L) LOEC: 37.2 ng/L (0.0372 µg/L)	Hamer et al. 1985b; summarized in U.S. EPA/EFED 1988
<i>Daphnia magna</i>	Chronic study (NOS)	NOEC: 0.00198 µg/L LOEC: 0.0035 µg/L	U.S. EPA/EFED 1994a,b
<i>Daphnia magna</i>	21-day study. See Table 1 of publication. No details.	NOEC: 0.002 µg/L. This appears to be the same study summarized in U.S. EPA/EFED 1994a,b.	Maund et al. 1998
<i>Mysidopsis bahia</i> (mysid shrimp)	Life-cycle study. DER summarizes information on measured concentrations for a complete listing of measured concentrations is not given.	NOEC for survival and weight: 1.7 ng/L (0.0017 µg/L) NOEC for reproduction: 0.22 ng/L (0.00022 µg/L) LOEC for reproduction: 0.46 ng/L (0.00046 µg/L)	Thompson 1987

Appendix 4: Toxicity to Aquatic Invertebrates (*continued*)

A4 Table 3: Field/Mesocosm Studies			
Species	Exposure	Response	Reference
Mixed invertebrates in 25 m ³ pond mesocosms. 1 m deep water with 15 cm of hydrosol. Macrophytes trimmed to allow 60% of water surface area to be clear for drift.	Deposition/drift treatment of 0.17 g a.i./ha and 1.7 g a.i./ha. Four applications are two week intervals. At lower rate, the concentration of lambda-cyhalothrin in water was about 0.002 µg/L. At high rate, peak water concentrations were about 0.023 µg/L to 0.094 µg/L. Concentrations declined rapidly after applications.	No adverse effects on zooplankton populations. Abnormal behavior in Notonectidae and Gyrrinidae immediately after applications. Significant decreases in amphipods (low and high rates) and isopods (at higher rate). Decrease in chironomids after initial application but recovery by third application. Increase in phytoplankton and periphyton associated with reduced grazing by invertebrates. LOEL for transient direct effects on invertebrates: 0.002 µg/L. 0.002 µg/L may be considered a marginal longer-term NOEC based on marginal and rare increases in phytoplankton and periphyton (indicative of marginal and rare effects on invertebrates).	Farmer et al. 1995
UK Mesocosms, 5x5 m, 1 m deep concrete mesocosms (algal, macrophyte, zooplankton and macroinvertebrates)	Low rate: 0.17 g a.i./ha, 4 sprays at 2 week intervals. Concentrations in water not reported.	Major effects on amphipods and Coleoptera without recovery by end of study. Major effects on isopods with recovery by end of study. No effects on other invertebrates, microorganisms, or algae.	Hamer et al. 1994
UK Mesocosms, 5x5 m, 1 m deep concrete mesocosms (algal, macrophyte, zooplankton and macroinvertebrates)	High rate: 1.7 g a.i./ha, 4 sprays at 2 week intervals. Concentrations in water not reported.	Major effects on amphipods, isopods, Ephemeroptera, and Coleoptera without recovery by end of study. No effects on other invertebrates, microorganisms, or algae.	Hamer et al. 1994
U.S. Mesocosms: 16 450 m ³ mesocosms. Mixed populations of pond-collected invertebrates, fish, and macrophytes.	Lambda-cyhalothrin applied in multiple times. High rate: Peak concentrations of about 0.09 to 0.1 µg/L. Mid rate: Peak concentration of about 0.01 µg/L. Low rate: Below the limit of detection (0.001 µg/L).	Low Rate: Minor and reversible effects in Baetidae (Ephemeroptera). Mid Rate: Significant adverse effects in some Hemiptera with no indication of recovery in Veliidae (water striders). Adverse effects with recovery in Trichoptera and some Diptera. High Rate: Adverse effects in many invertebrates. The most sensitive group appears to be Gerridae (water striders). Transient LOEC: 0.001 µg/L. LOEC: 0.01 µg/L.	Hill et al. 1994 (also summarized in Hamer et al. 1994)

Appendix 4: Toxicity to Aquatic Invertebrates (continued)

A4 Table 3: Field/Mesocosm Studies			
Species	Exposure	Response	Reference
Mixed invertebrates in natural stream – not a mesocosm.	2 pulse (30 min) treatments: first pulse 0.10, 1.00, and 10.0 µg/L second pulse: 0.05, 0.50, and 5.00 µg/L	Increase in macroinvertebrate drift. Significant change in community structure only at highest exposures – 10 and 5 µg/L. Recovery within 2 weeks. Transient LOEC: 0.1/0.05 µg/L Community NOEC: 1/0.5 µg/L	Heckmann and Friberg 2005
Mixed invertebrates in outdoor experimental channel (mesocosm)	Pulse concentrations 0.001, 0.01, 0.1, and 1.0 µg/L.	0.001 µg/L: significant increase in <i>Gammarus</i> drift but no effect on two insect species – i.e. <i>Baetis rhodani</i> (horse fly) and <i>Leuctra fusca/digitata</i> (stonefly). 0.01 µg/L: increased drift in all species. Transient LOEC: 0.001 µg/L	Lauridsen and Friberg 2005
Mixed community of <i>Gammarus pulex</i> , <i>Leuctra nigra</i> , <i>Heptagenia sulphurea</i> and <i>Ancylus fluviatilis</i>	Outdoor stream mesocosm. 10.65 or 106.5 ng/L for 90 min in the laboratory and after 24 h introduced to the experimental stream channels.	Higher algal biomass and decrease in litter decomposition at higher concentration relative to lower concentration and controls. NOEC: 0.011 µg/L.	Rasmussen et al. 2008
Mixed populations in ditch mesocosms	3 applications at one week intervals at 10, 25, 50, 100, and 250 ng/L.	Transient toxicity at 0.01 µg/L. Greater toxicity and lower recovery at higher concentrations. Effects persisted somewhat longer in macrophyte-dominated ditches (see Fig. 3 of study). At 0.25 µg/L, effects in macrophyte-dominated ditches persisted to 45 days.	Roessink et al. 2005; also summarized in Roessink et al. 2008
Experimental ditches with mixed aquatic invertebrates with either macrophyte or plankton dominated plant communities.	Concentrations of 0, 10, 25, 50, 100, 250 ng/L.	NOEC for community response of 10 ng/L. At this concentration, however, adverse effects were noted on <i>Chaoborus obscuripes</i> (midge). Transient LOEC: 0.01 µg/L.	Schroer et al. 2004

Appendix 4: Toxicity to Aquatic Invertebrates (*continued*)

A4 Table 3: Field/Mesocosm Studies			
Species	Exposure	Response	Reference
Drainage ditch ecosystem: macrophyte-dominated with macroinvertebrates, zooplankton, and Phytoplankton.	Single application of 0, 10, 25, 50, 100, 250 ng active ingredient a.i./L in spring and late summer	<p>No remarkable differences between application times. Most sensitive species were aquatic invertebrates <i>Chaoborus obscuripes</i> (midge) and <i>Gammarus pulex</i> (amphipod) with NOEC <10 ng/L (<0.01 µg/L). Measured concentrations were only about 70%-90% of nominal at one hour after application.</p> <p>Recovery of <i>Chaoborus obscuripes</i> and other species within 3 weeks.</p> <p>Decreases in litter decomposition at two higher concentrations.</p> <p>Transient LOEC: 0.01 µg/L.</p>	Van Wijngaarden et al. 2006

Appendix 5: Generic Gleams-Driver Simulations, Single Application

Table 1: Effective Offsite Application Rate (lb/acre)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.000056 (0 - 0.00101)	0 (0 - 0.00064)	0 (0 - 9.50E-06)
Dry and Temperate Location	2.34E-05 (1.37E-06 - 0.00015)	1.83E-07 (0 - 0.000098)	0 (0 - 1.42E-05)
Dry and Cold Location	0.000006 (4.70E-08 - 0.000082)	0 (0 - 5.60E-06)	0 (0 - 0)
Average Rainfall and Warm Location	0.00143 (0.00063 - 0.0044)	0.00102 (0.000236 - 0.0039)	0.000063 (0 - 0.00064)
Average Rainfall and Temperate Location	0.00121 (0.000308 - 0.0047)	0.00079 (0.000102 - 0.0047)	1.96E-05 (0 - 0.00085)
Average Rainfall and Cool Location	0.00066 (0.000233 - 0.00166)	0.000268 (0.000042 - 0.00123)	0 (0 - 0.000132)
Wet and Warm Location	0.00185 (0.00066 - 0.0077)	0.0013 (0.00048 - 0.009)	0.000157 (2.07E-05 - 0.0029)
Wet and Temperate Location	0.0008 (0.00049 - 0.00217)	0.00057 (0.000284 - 0.00181)	0.000047 (9.80E-06 - 0.00037)
Wet and Cool Location	0.0039 (0.00219 - 0.0063)	0.00203 (0.00085 - 0.0055)	0.00014 (1.08E-05 - 0.0012)
Average of Central Values:			0.000605
25th Percentile of Lower Bounds:			0
Maximum Value:			0.009
Summary of Values:			0.00061 (0 - 0.009)

Appendix 5: Generic Gleams-Driver Simulations, Single Application *(continued)*

L-C Run 02			
Table 2: Concentration in Top 12 Inches of Soil (ppm)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.173 (0.172 - 0.173)	0.158 (0.157 - 0.158)	0.158 (0.157 - 0.158)
Dry and Temperate Location	0.173 (0.172 - 0.173)	0.157 (0.157 - 0.158)	0.157 (0.157 - 0.158)
Dry and Cold Location	0.172 (0.171 - 0.173)	0.157 (0.156 - 0.157)	0.157 (0.156 - 0.157)
Average Rainfall and Warm Location	0.172 (0.171 - 0.173)	0.157 (0.156 - 0.157)	0.157 (0.156 - 0.157)
Average Rainfall and Temperate Location	0.172 (0.171 - 0.172)	0.157 (0.156 - 0.157)	0.157 (0.156 - 0.157)
Average Rainfall and Cool Location	0.172 (0.171 - 0.172)	0.157 (0.156 - 0.157)	0.157 (0.156 - 0.157)
Wet and Warm Location	0.171 (0.171 - 0.172)	0.156 (0.156 - 0.157)	0.156 (0.156 - 0.157)
Wet and Temperate Location	0.172 (0.171 - 0.172)	0.157 (0.156 - 0.157)	0.157 (0.156 - 0.157)
Wet and Cool Location	0.171 (0.171 - 0.172)	0.156 (0.156 - 0.157)	0.156 (0.156 - 0.157)
Average of Central Values:			0.1619
25th Percentile of Lower Bounds:			0.156
Maximum Value:			0.173
Summary of Values:			0.162 (0.156 - 0.173)

Appendix 5: Generic Gleams-Driver Simulations, Single Application *(continued)*

L-C Run 02			
Table 3: Concentration in Top 60 Inches of Soil (ppm)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.035 (0.034 - 0.035)	0.0315 (0.0314 - 0.0316)	0.0315 (0.0314 - 0.0315)
Dry and Temperate Location	0.035 (0.034 - 0.035)	0.0315 (0.0313 - 0.0315)	0.0315 (0.0313 - 0.0315)
Dry and Cold Location	0.034 (0.034 - 0.035)	0.0314 (0.0312 - 0.0315)	0.0314 (0.0312 - 0.0315)
Average Rainfall and Warm Location	0.034 (0.034 - 0.035)	0.0314 (0.0313 - 0.0315)	0.0314 (0.0313 - 0.0315)
Average Rainfall and Temperate Location	0.034 (0.034 - 0.034)	0.0314 (0.0312 - 0.0314)	0.0314 (0.0312 - 0.0314)
Average Rainfall and Cool Location	0.034 (0.034 - 0.034)	0.0314 (0.0312 - 0.0314)	0.0314 (0.0312 - 0.0314)
Wet and Warm Location	0.034 (0.034 - 0.034)	0.0312 (0.0312 - 0.0314)	0.0312 (0.0312 - 0.0314)
Wet and Temperate Location	0.034 (0.034 - 0.034)	0.0314 (0.0312 - 0.0314)	0.0314 (0.0312 - 0.0314)
Wet and Cool Location	0.034 (0.034 - 0.034)	0.0312 (0.0312 - 0.0314)	0.0312 (0.0312 - 0.0314)
Average of Central Values:			0.0323
25th Percentile of Lower Bounds:			0.0312
Maximum Value:			0.035
Summary of Values:			0.032 (0.0312 - 0.035)

Appendix 5: Generic Gleams-Driver Simulations, Single Application *(continued)*

L-C Run 02			
Table 4: Maximum Penetration into Soil Column (inches)			
Site	Clay	Loam	Sand
Dry and Warm Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 4)
Dry and Temperate Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 4)
Dry and Cold Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 4)
Average Rainfall and Warm Location	4 (4 - 4)	4 (4 - 4)	8 (4 - 8)
Average Rainfall and Temperate Location	4 (4 - 4)	4 (4 - 4)	8 (4 - 8)
Average Rainfall and Cool Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 8)
Wet and Warm Location	4 (4 - 4)	4 (4 - 4)	8 (8 - 8)
Wet and Temperate Location	4 (4 - 4)	4 (4 - 4)	8 (4 - 8)
Wet and Cool Location	4 (4 - 4)	4 (4 - 8)	8 (8 - 8)
Average of Central Values:			4.74
25th Percentile of Lower Bounds:			4
Maximum Value:			8
Summary of Values:			4.74 (4 - 8)

Appendix 5: Generic Gleams-Driver Simulations, Single Application (continued)

L-C Run 02			
Table 5: Stream, Maximum Peak Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.005 (4E-17 - 0.07)	4E-105 (0 - 0.08)	0 (0 - 0.0025)
Dry and Temperate Location	0.0019 (0.00014 - 0.013)	0.00005 (0 - 0.01)	0 (0 - 0.0021)
Dry and Cold Location	0.0007 (0.000005 - 0.008)	0 (0 - 0.0008)	0 (0 - 0)
Average Rainfall and Warm Location	0.08 (0.028 - 0.23)	0.08 (0.021 - 0.29)	0.014 (4.0E-10 - 0.16)
Average Rainfall and Temperate Location	0.06 (0.018 - 0.3)	0.06 (0.012 - 0.4)	0.004 (1.7E-115 - 0.22)
Average Rainfall and Cool Location	0.04 (0.01 - 0.13)	0.024 (0.004 - 0.14)	1.3E-47 (0 - 0.04)
Wet and Warm Location	0.09 (0.03 - 0.6)	0.1 (0.03 - 1.03)	0.029 (0.004 - 0.7)
Wet and Temperate Location	0.04 (0.015 - 0.1)	0.04 (0.013 - 0.13)	0.008 (0.0014 - 0.06)
Wet and Cool Location	0.08 (0.04 - 0.4)	0.09 (0.029 - 0.5)	0.02 (0.0017 - 0.22)
Average of Central Values:			0.0321
25th Percentile of Lower Bounds:			8.5E-116
Maximum Value:			1.03
Summary of Values:			0.032 (8.5E-116 - 1.03)

Appendix 5: Generic Gleams-Driver Simulations, Single Application (continued)

L-C Run 02			
Table 6: Stream, Annual Average Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.00007 (2.6E-19 - 0.0008)	2.3E-107 (0 - 0.0007)	0 (0 - 0.00002)
Dry and Temperate Location	0.000032 (1.3E-06 - 0.00021)	5.0E-07 (0 - 0.0001)	0 (0 - 0.000026)
Dry and Cold Location	0.000007 (7.0E-08 - 0.00008)	0 (0 - 0.000007)	0 (0 - 0)
Average Rainfall and Warm Location	0.002 (0.001 - 0.004)	0.0014 (0.0005 - 0.004)	0.00013 (2.3E-12 - 0.0013)
Average Rainfall and Temperate Location	0.0015 (0.0005 - 0.005)	0.0009 (0.00019 - 0.005)	0.00004 (1E-117 - 0.0016)
Average Rainfall and Cool Location	0.0009 (0.0003 - 0.0018)	0.0003 (0.00005 - 0.0013)	8E-50 (0 - 0.00023)
Wet and Warm Location	0.0024 (0.0014 - 0.006)	0.002 (0.0009 - 0.009)	0.0004 (0.00004 - 0.005)
Wet and Temperate Location	0.0014 (0.0009 - 0.0024)	0.0009 (0.0005 - 0.0021)	0.0001 (0.000021 - 0.0005)
Wet and Cool Location	0.0031 (0.0021 - 0.005)	0.0021 (0.001 - 0.005)	0.00023 (0.000019 - 0.002)
Average of Central Values:			0.000737
25th Percentile of Lower Bounds:			5E-118
Maximum Value:			0.009
Summary of Values:			0.00074 (5E-118 - 0.009)

Appendix 5: Generic Gleams-Driver Simulations, Single Application (continued)

L-C Run 02			
Table 7: Pond, Maximum Peak Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.0007 (0 - 0.011)	0 (0 - 0.011)	0 (0 - 0.0004)
Dry and Temperate Location	0.00027 (0.000016 - 0.0017)	0.000003 (0 - 0.0015)	0 (0 - 0.0006)
Dry and Cold Location	0.00008 (7.0E-07 - 0.0012)	0 (0 - 0.0001)	0 (0 - 0)
Average Rainfall and Warm Location	0.013 (0.004 - 0.03)	0.013 (0.003 - 0.04)	0.0023 (0 - 0.021)
Average Rainfall and Temperate Location	0.011 (0.0027 - 0.05)	0.01 (0.0018 - 0.06)	0.0007 (0 - 0.03)
Average Rainfall and Cool Location	0.006 (0.0016 - 0.019)	0.004 (0.0007 - 0.022)	0 (0 - 0.005)
Wet and Warm Location	0.014 (0.005 - 0.05)	0.015 (0.005 - 0.09)	0.004 (0.0006 - 0.06)
Wet and Temperate Location	0.006 (0.0025 - 0.017)	0.006 (0.0021 - 0.021)	0.0013 (0.00025 - 0.009)
Wet and Cool Location	0.013 (0.008 - 0.03)	0.014 (0.006 - 0.05)	0.004 (0.00029 - 0.025)
Average of Central Values:			0.00512
25th Percentile of Lower Bounds:			0
Maximum Value:			0.09
Summary of Values:			0.0051 (0 - 0.09)

Appendix 5: Generic Gleams-Driver Simulations, Single Application (continued)

L-C Run 02			
Table 8: Pond, Annual Average Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.000008 (0 - 0.0001)	0 (0 - 0.00008)	0 (0 - 2.9E-06)
Dry and Temperate Location	0.000003 (1.5E-07 - 0.000029)	2.3E-08 (0 - 0.000013)	0 (0 - 0.000004)
Dry and Cold Location	7.0E-07 (8.0E-09 - 0.000009)	0 (0 - 7.0E-07)	0 (0 - 0)
Average Rainfall and Warm Location	0.00028 (0.00014 - 0.0007)	0.00021 (0.00007 - 0.0007)	0.000023 (0 - 0.00021)
Average Rainfall and Temperate Location	0.00022 (0.00006 - 0.0007)	0.00014 (0.000026 - 0.0007)	0.000007 (0 - 0.00028)
Average Rainfall and Cool Location	0.00012 (0.00004 - 0.00028)	0.00004 (0.000006 - 0.0002)	0 (0 - 0.00004)
Wet and Warm Location	0.0004 (0.0002 - 0.0008)	0.0003 (0.00013 - 0.0011)	0.00006 (0.000007 - 0.0007)
Wet and Temperate Location	0.0002 (0.00014 - 0.0003)	0.00014 (0.00008 - 0.00031)	0.000018 (0.000005 - 0.0001)
Wet and Cool Location	0.0006 (0.0004 - 0.0008)	0.0004 (0.00017 - 0.0007)	0.00004 (0.000004 - 0.00024)
Average of Central Values:			0.0001189
25th Percentile of Lower Bounds:			0
Maximum Value:			0.0011
Summary of Values:			0.000119 (0 - 0.0011)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Appendix 6: Generic Gleams-Driver Simulations, six applications at 2 week intervals

Table 1: Effective Offsite Application Rate (lb/acre)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.00047 (0 - 0.0048)	0 (0 - 0.0037)	0 (0 - 0.000053)
Dry and Temperate Location	0.000154 (9.60E-06 - 0.00093)	1.29E-06 (0 - 0.0007)	0 (0 - 0.000094)
Dry and Cold Location	0.000104 (2.58E-05 - 0.00039)	0 (0 - 1.97E-05)	0 (0 - 0)
Average Rainfall and Warm Location	0.0078 (0.0039 - 0.0172)	0.0057 (0.00149 - 0.0154)	0.0004 (0 - 0.00284)
Average Rainfall and Temperate Location	0.0067 (0.0023 - 0.0165)	0.0041 (0.00088 - 0.0166)	0.000117 (0 - 0.00312)
Average Rainfall and Cool Location	0.0032 (0.00157 - 0.0111)	0.00135 (0.000287 - 0.0088)	0 (0 - 0.00066)
Wet and Warm Location	0.0104 (0.0061 - 0.049)	0.0086 (0.0035 - 0.071)	0.00098 (0.000145 - 0.0174)
Wet and Temperate Location	0.0056 (0.0035 - 0.0144)	0.0041 (0.00204 - 0.0131)	0.00033 (0.000068 - 0.00265)
Wet and Cool Location	0.0171 (0.0104 - 0.0304)	0.0104 (0.0043 - 0.0307)	0.00072 (0.000125 - 0.0044)
Average of Central Values:			0.00327
25th Percentile of Lower Bounds:			0
Maximum Value:			0.071
Summary of Values:			0.0033 (0 - 0.071)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 2: Concentration in Top 12 Inches of Soil (ppm)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.66 (0.63 - 0.74)	0.6 (0.58 - 0.68)	0.62 (0.59 - 0.65)
Dry and Temperate Location	0.65 (0.62 - 0.75)	0.59 (0.57 - 0.69)	0.61 (0.58 - 0.78)
Dry and Cold Location	0.61 (0.59 - 0.64)	0.55 (0.53 - 0.58)	0.57 (0.55 - 0.61)
Average Rainfall and Warm Location	0.62 (0.6 - 0.64)	0.57 (0.55 - 0.58)	0.58 (0.57 - 0.61)
Average Rainfall and Temperate Location	0.63 (0.61 - 0.64)	0.57 (0.56 - 0.59)	0.59 (0.57 - 0.6)
Average Rainfall and Cool Location	0.62 (0.61 - 0.64)	0.56 (0.54 - 0.58)	0.58 (0.57 - 0.6)
Wet and Warm Location	0.57 (0.55 - 0.59)	0.52 (0.5 - 0.53)	0.54 (0.53 - 0.55)
Wet and Temperate Location	0.62 (0.61 - 0.67)	0.56 (0.54 - 0.61)	0.58 (0.56 - 0.62)
Wet and Cool Location	0.57 (0.56 - 0.59)	0.52 (0.5 - 0.53)	0.54 (0.52 - 0.55)
Average of Central Values:			0.585
25th Percentile of Lower Bounds:			0.545
Maximum Value:			0.78
Summary of Values:			0.59 (0.545 - 0.78)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 3: Concentration in Top 60 Inches of Soil (ppm)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.132 (0.127 - 0.148)	0.121 (0.115 - 0.136)	0.124 (0.118 - 0.131)
Dry and Temperate Location	0.13 (0.124 - 0.15)	0.119 (0.113 - 0.139)	0.122 (0.116 - 0.155)
Dry and Cold Location	0.122 (0.118 - 0.127)	0.11 (0.106 - 0.116)	0.114 (0.111 - 0.122)
Average Rainfall and Warm Location	0.124 (0.121 - 0.127)	0.113 (0.111 - 0.117)	0.117 (0.114 - 0.122)
Average Rainfall and Temperate Location	0.125 (0.123 - 0.129)	0.114 (0.112 - 0.118)	0.117 (0.115 - 0.12)
Average Rainfall and Cool Location	0.125 (0.122 - 0.128)	0.112 (0.109 - 0.115)	0.116 (0.113 - 0.12)
Wet and Warm Location	0.114 (0.111 - 0.117)	0.103 (0.101 - 0.106)	0.108 (0.105 - 0.11)
Wet and Temperate Location	0.124 (0.121 - 0.134)	0.112 (0.109 - 0.122)	0.115 (0.112 - 0.124)
Wet and Cool Location	0.114 (0.111 - 0.118)	0.103 (0.101 - 0.106)	0.107 (0.104 - 0.109)
Average of Central Values:			0.1169
25th Percentile of Lower Bounds:			0.11
Maximum Value:			0.155
Summary of Values:			0.117 (0.11 - 0.155)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 4: Maximum Penetration into Soil Column (inches)			
Site	Clay	Loam	Sand
Dry and Warm Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 8)
Dry and Temperate Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 8)
Dry and Cold Location	4 (4 - 4)	4 (4 - 4)	4 (4 - 4)
Average Rainfall and Warm Location	4 (4 - 4)	8 (4 - 8)	8 (8 - 8)
Average Rainfall and Temperate Location	4 (4 - 4)	4 (4 - 8)	8 (8 - 8)
Average Rainfall and Cool Location	4 (4 - 4)	4 (4 - 8)	8 (4 - 8)
Wet and Warm Location	4 (4 - 4)	8 (4 - 8)	8 (8 - 8)
Wet and Temperate Location	4 (4 - 4)	4 (4 - 8)	8 (8 - 8)
Wet and Cool Location	4 (4 - 8)	8 (8 - 8)	8 (8 - 8)
Average of Central Values:			5.33
25th Percentile of Lower Bounds:			4
Maximum Value:			8
Summary of Values:			5.33 (4 - 8)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 5: Stream, Maximum Peak Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.04 (0 - 0.5)	0 (0 - 0.5)	0 (0 - 0.014)
Dry and Temperate Location	0.013 (0.001 - 0.07)	0.00015 (0 - 0.07)	0 (0 - 0.014)
Dry and Cold Location	0.008 (0.0023 - 0.027)	0 (0 - 0.0027)	0 (0 - 0)
Average Rainfall and Warm Location	0.4 (0.15 - 0.9)	0.4 (0.14 - 1.24)	0.09 (0 - 0.6)
Average Rainfall and Temperate Location	0.3 (0.13 - 1.07)	0.3 (0.07 - 1.51)	0.026 (0 - 0.8)
Average Rainfall and Cool Location	0.16 (0.06 - 0.7)	0.13 (0.03 - 0.9)	0 (0 - 0.18)
Wet and Warm Location	0.5 (0.19 - 4.2)	0.6 (0.21 - 7.9)	0.2 (0.029 - 3.4)
Wet and Temperate Location	0.21 (0.1 - 0.6)	0.24 (0.1 - 0.9)	0.05 (0.01 - 0.5)
Wet and Cool Location	0.4 (0.17 - 1.3)	0.5 (0.15 - 1.87)	0.09 (0.016 - 0.7)
Average of Central Values:			0.1725
25th Percentile of Lower Bounds:			0
Maximum Value:			7.9
Summary of Values:			0.172 (0 - 7.9)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 6: Stream, Annual Average Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.0005 (0 - 0.004)	0 (0 - 0.004)	0 (0 - 0.00011)
Dry and Temperate Location	0.00021 (0.000009 - 0.0014)	1.4E-06 (0 - 0.0007)	0 (0 - 0.00017)
Dry and Cold Location	0.00012 (0.000031 - 0.0004)	0 (0 - 0.000023)	0 (0 - 0)
Average Rainfall and Warm Location	0.012 (0.006 - 0.021)	0.008 (0.003 - 0.023)	0.0008 (0 - 0.006)
Average Rainfall and Temperate Location	0.008 (0.004 - 0.024)	0.006 (0.0014 - 0.026)	0.00024 (0 - 0.006)
Average Rainfall and Cool Location	0.005 (0.0022 - 0.01)	0.0017 (0.0004 - 0.008)	0 (0 - 0.0012)
Wet and Warm Location	0.016 (0.01 - 0.04)	0.014 (0.007 - 0.07)	0.0026 (0.00031 - 0.027)
Wet and Temperate Location	0.009 (0.006 - 0.015)	0.007 (0.004 - 0.014)	0.0007 (0.00014 - 0.004)
Wet and Cool Location	0.017 (0.012 - 0.024)	0.012 (0.006 - 0.024)	0.0011 (0.00017 - 0.005)
Average of Central Values:			0.00452
25th Percentile of Lower Bounds:			0
Maximum Value:			0.07
Summary of Values:			0.0045 (0 - 0.07)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 7: Pond, Maximum Peak Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.006 (0 - 0.07)	0 (0 - 0.07)	0 (0 - 0.0022)
Dry and Temperate Location	0.0019 (0.00011 - 0.01)	0.000023 (0 - 0.011)	0 (0 - 0.004)
Dry and Cold Location	0.0012 (0.00026 - 0.004)	0 (0 - 0.0004)	0 (0 - 0)
Average Rainfall and Warm Location	0.06 (0.026 - 0.14)	0.07 (0.022 - 0.21)	0.013 (0 - 0.11)
Average Rainfall and Temperate Location	0.05 (0.02 - 0.16)	0.05 (0.013 - 0.22)	0.004 (0 - 0.11)
Average Rainfall and Cool Location	0.025 (0.011 - 0.1)	0.019 (0.005 - 0.13)	0 (0 - 0.027)
Wet and Warm Location	0.07 (0.032 - 0.4)	0.09 (0.03 - 0.7)	0.029 (0.005 - 0.4)
Wet and Temperate Location	0.04 (0.014 - 0.09)	0.04 (0.015 - 0.12)	0.008 (0.0017 - 0.07)
Wet and Cool Location	0.06 (0.028 - 0.14)	0.07 (0.024 - 0.2)	0.015 (0.0022 - 0.11)
Average of Central Values:			0.02675
25th Percentile of Lower Bounds:			0
Maximum Value:			0.7
Summary of Values:			0.0267 (0 - 0.7)

Appendix 6: Generic Gleams-Driver Simulation, six applications at 2 week intervals
(continued)

Table 8: Pond, Annual Average Concentration in Surface Water (ug/L or ppb)			
Site	Clay	Loam	Sand
Dry and Warm Location	0.00006 (0 - 0.0005)	0 (0 - 0.0005)	0 (0 - 0.000016)
Dry and Temperate Location	0.000024 (0.000001 - 0.00018)	1.6E-07 (0 - 0.00009)	0 (0 - 0.000029)
Dry and Cold Location	0.000013 (0.000003 - 0.00005)	0 (0 - 2.5E-06)	0 (0 - 0)
Average Rainfall and Warm Location	0.0017 (0.0009 - 0.004)	0.0013 (0.0005 - 0.004)	0.00015 (0 - 0.0011)
Average Rainfall and Temperate Location	0.0012 (0.0005 - 0.003)	0.0009 (0.00019 - 0.004)	0.00004 (0 - 0.0011)
Average Rainfall and Cool Location	0.0006 (0.00029 - 0.0014)	0.00024 (0.00006 - 0.0012)	0 (0 - 0.0002)
Wet and Warm Location	0.0024 (0.0015 - 0.005)	0.0021 (0.001 - 0.006)	0.0004 (0.00006 - 0.003)
Wet and Temperate Location	0.0014 (0.001 - 0.0022)	0.001 (0.0006 - 0.002)	0.00013 (0.00003 - 0.0007)
Wet and Cool Location	0.0029 (0.002 - 0.004)	0.002 (0.0011 - 0.004)	0.00021 (0.00004 - 0.0009)
Average of Central Values:			0.000695
25th Percentile of Lower Bounds:			0
Maximum Value:			0.006
Summary of Values:			0.0007 (0 - 0.006)