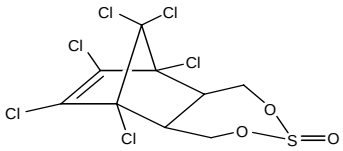
Office of Prevention, Pesticides,
and Toxic Substances

Appendix 1 to 2007 Addendum: Environmental Fate and Ecological Risk Assessment of Endosulfan



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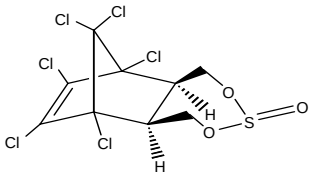

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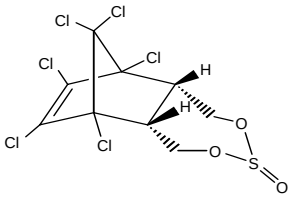

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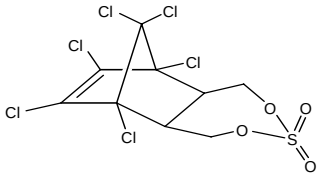

endosulfan sulfate

Prepared by:
Keith Sappington, Senior Biologist, ERB V
Faruque Khan, Senior Fate Scientist, ERB I

Contributor:
Kristina Garber, Biologist, ERB IV

Reviewed by:
Tom Steeger, Senior Biologist, ERB IV

Approved by:
Mah Shamim, Chief, ERB V

United States Environmental Protection Agency
Office of Pesticide Programs
Environmental Fate and Effects Division
Environmental Risk Branch V
1200 Pennsylvania Ave.
Mail Code 7507P
Washington, D.C. 20460

1. INTRODUCTION

This appendix contains detailed information to support the 2007 Addendum to the 2002 environmental fate and ecological risk assessment (ERA) chapter in support of the reregistration eligibility decision (RED) for endosulfan (Memo dated October 31, 2007, DP Barcode D346213). As discussed in the 2007 Addendum, new information related to endosulfan toxicity, bioaccumulation, monitoring and transport, and ecological incidence have been obtained by the Office of Pesticide Programs (OPP). The purpose of the 2007 Addendum is to conduct a preliminary assessment of this new information in order to: (1) address the extent to which the previous ecological risk assessment for endosulfan might change in relation to this new information, and (2) indicate future avenues where additional data analysis and risk characterization endosulfan are needed. This assessment is considered preliminary because a complete review of the new data has not been completed and therefore, decisions regarding the acceptability and utility of some data might change, pending further review. This appendix addendum is organized according to new information on endosulfan related to the following assessment topics:

- Section 2: Bioaccumulation
- Section 3: Ecological Effects
- Section 4: Ecological Exposure
- Section 5: Monitoring and Long Range Transport
- Section 6: Risk Characterization

Detailed information pertaining to new information on bioaccumulation and ecological effects are found in Attachments A through D.

2. NEW DATA ON BIOACCUMULATION

A preliminary review and analysis of endosulfan bioaccumulation data is summarized in this section. The purpose of this review is to indicate how the Agency's understanding of the bioaccumulation potential of endosulfan (and sulfate metabolite) might change as a result of additional information being considered since the publication of EPA's 2002 ERA for endosulfan. This review is considered preliminary for two reasons. First, it is not intended to be comprehensive. Specifically, the literature review of empirical bioaccumulation studies focused on controlled experiments of endosulfan bioconcentration or bioaccumulation rather than uncontrolled field studies on the distribution of endosulfan in various environmental compartments. The scope was constrained in this way primarily because of practical limitations (time constraints) and also the expectation that biomagnification of endosulfan (and degradates) in aquatic food webs would not likely be a major factor given its moderate hydrophobicity (log K_{ow} 3-4.5). Controlled laboratory studies of bioconcentration generally involve less uncertainty in quantifying chemical exposure by organisms and thus, generally contain less uncertainty in calculated BCFs compared to field studies. Second, the available data were not subjected to formal data evaluation procedures (e.g., Data Evaluation Records), again, due to time and resource constraints.

Given these caveats, the following review of new information regarding endosulfan bioaccumulation is focused on two areas: (1) synthesizing results from empirical bioaccumulation studies, and (2) addressing key bioaccumulation assessment issues through the use of food web bioaccumulation models. Findings from the review of empirical bioaccumulation studies are provided in Section 2.1 with supporting information provided in Attachment A. Similarly, findings from the consideration of bioaccumulation food web modeling are provided in Section 2.2 with supporting information placed in Attachment B.

2.1 Findings from Empirical Bioaccumulation Studies

2.1.1. Bioconcentration/Bioaccumulation by Fish

Bioconcentration data were identified and reviewed for seven species of fish, including sheepshead minnow (*Cyprinodon variegatus)*, zebra fish (*Brachydanio rerio)*, yellow tetra *(Hyphessobrycon bifasciatus),* striped mullet *(Mugil cephalus),* pinfish (*Lagodon rhomboids),* long whiskers catfish (*Mystus gulio),* and spot *(Leiostomus xanthurus;* Table 2-1). The reported BCF values for fish ranged from approximately 20 to 11,600 (L/kg wet wt.). With the exception of one species (yellow tetra), BCFs were less than 3,000 for the remaining six fish species. As discussed in Attachment A, the kinetic-based BCF for yellow tetra appears inconsistent with the observed accumulation pattern reported in this study and therefore, is considered highly uncertain. On the basis of observed residues in tissue and calculated (nominal) concentrations in water, a ratio-based BCF of 5,670 can be calculated from the study with yellow tetra. This ratio-based BCF value also contains considerable uncertainty because it is based on a static-renewal exposure system and concentrations in test solution were not verified analytically. An evaluation of the fish BCF data quality indicates most of the BCF values have significant limitations because none of the BCF studies satisfied all three screening criteria (documentation of steady-state conditions, measurement and stability of exposure concentrations, and quantification of parent and metabolite compounds). Based on these screening criteria, BCF values for fish from the highest quality studies appear to be in the 1000 to 3000 range (Hansen and Cripe, 1991 for sheepshead minnow and Schimmel et al., 1977 for striped mullet, Table 2-1). No studies involving endosulfan accumulation from multiple exposure routes (i.e., bioaccumulation) were identified for fish. However, as noted previously, this review focused on controlled laboratory studies of endosulfan bioaccumulation rather than field studies and thus, appropriate bioaccumulation data may not have been identified.

2.1.2 Bioconcentration/Bioaccumulation by Invertebrates

Bioconcentration studies with aquatic invertebrates were available for five species of invertebrates and included the blue mussel (*Mytilus edulis),* grass shrimp, (*Palaemonetes pugio),* oyster, (*Crassostrea madrasensis)*, clam, (*Katelysia opima)* and red swamp crayfish, (*Procambarus clarkii).* Based on the studies presented in Table 2-1, the bioconcentration of endosulfan in aquatic invertebrates appears to be lower than those reported for fish, ranging from about 20 to 600 (L/kg w.w.). The value of 1.9 from Naqvi

and Newton (1990) for crayfish is considered highly suspect and is not discussed further (see Attachment A). Bioaccumulation studies (i.e., those that included exposure to multiple uptake routes) were available for three invertebrates, including the mussel (*M. galloprovincialis)*, eastern oyster, (*C. virginica)*, and the water flea, (*Daphnia magna*; Table 2-2). Bioaccumulation factors (Table 2-2) for the eastern oyster and *D. magna* for total endosulfan are approximately 600. In a short-term study by DeLorenzo et al (2002), uptake of endosulfan from food (contaminated algae) by *D. magna* was documented as negligible compared to uptake from the water column.

2.1.3 Depuration Half Life

The depuration of endosulfan and endosulfan sulfate by fish appears to be relatively rapid, with half lives ranging from 2-6 days for zebra fish, yellow tetra, and striped mullet (Toledo and Jonsson, 1992; Jonsson and Toledo, 1993; Schimmel et al., 1977; Attachment A). It is noted that in two studies, calculated half lives in fish (approx. 2 days) appear inconsistent with observed accumulation in tissue (i.e., steady-state accumulation was not observed after 21 and 28 days in yellow tetra and striped mullet, respectively when in theory, it should have been reached by 7 days based on depuration rates for these two species; Jonsson and Toledo, 1993; Schimmel et al., 1977). This inconsistency suggests that endosulfan accumulation by fish might be more complex than the assumption of simple first order kinetics, at least in some cases.

Information on the depuration of endosulfan by invertebrates was only available for the blue mussel, *M. edulis*. In one study, a depuration half life of 33.8 hours (about 1.5 days) was reported for blue mussel (Ernst, 1977), while a second long-term study suggested a depuration half-life on the order of two weeks for this species (Roberts, 1972). As noted in Attachment A, these two studies have a number of limitations which suggest these depuration half lives are uncertain and should be used with caution.

Table 2-1. Summary of Aquatic Bioconcentration Studies with Endosulfan

Chemical (formulation/ % ai) (*1)	Species	Study Design (*2)	Exposure Duration (Exposure Conc. µg/L)	BCF Method (SS) (*3)	Avg. BCF/ (BAF)	Range [SD] BCF/ (BAF)	N	Reference
Endosulfan 64% α / 36% β (TG/ 98%)	Sheepshead minnow *(Cyprinodon variegatus)*	FT / M / WB	28 d (5 levels, ~0.05-5.5)	Ratio, α+ β (SS NR)	1146 (*4)	318-2963	9	Hansen & Cripe (1991)
Endosulfan 2:1 α / β (TG/97%)	Zebra Fish *(Brachydanio rerio)*	SR / U / WB	21 d (1 level, 0.3)	Kinetic, α+ β+ sulfate	2650	[441]	3	Toledo and Jonsson (1992)
Endosulfan 2:1 α / β (TG/97%)	Yellow Tetra *(Hyphessobrycon bifasciatus)*	SR / U / WB	21 d (1 level, 0.3)	Kinetic, α+β+ sulfate Ratio	11583 (*5) 5670	[2361] ---	3 3	Jonsson and Toledo (1993)
endosulfan + 6 organochlorine pesticides (NR)	Blue Mussel *(Mytilus edulis)*	S / M / WB	7 d (1 level, 2.1➔0.14)	Ratio (SS assumed)	600	NR	NR	Ernst (1977) (*6)
Endosulfan 70% α / 30% β (TG, ai NR)	Striped mullet *(Mugil cephalus)*	FT / M / WB	28-d (1 level, 0.035 ± 0.006)	Ratio, α+ β+ sulfate (non-SS?)	2,755	NR	5	Schimmel et al (1977) (*6)
	Striped Mullet *(Mugil cephalus)*	FT / M / WB	96-h (3 levels, 0.36-0.49)	Ratio, α+ β+ sulfate (non-SS)	1115	1000-1344	3	
	Spot *(Leiostomus xanthurus)*	FT / M / WB	96-h (3 levels, 0.05-0.31)	Ratio, α+ β+ sulfate (SS NR)	780	620-895	3	
	Grass shrimp *(Palaemonetes pugio)*	FT / M / WB	96-h (5 levels, 0.16-1.75)	Ratio, α+ β+ sulfate (SS NR)	175	81-245	5	
	Pinfish (Lagodon rhomboids)	FT / M / WB	96-h (2 levels, 0.15-0.26)	Ratio, α+ β+ sulfate (SS NR)	1173	1046-1299	2	
Endosulfan (NR)	Blue Mussel *(Mytilus edulis)*	FT / U / WB	122-d (3 levels, 100-1000)	Ratio, α+ β (non-SS?)	12	8-17	3	Roberts (1972)
Endosulfan (NR)	Striped mullet *(Mugil cephalus)*	FT / M / Muscle	10-d (3 levels, 0.13- 1.25)	Ratio (SS NR)	18.4	18.1-18.6	3	Rajendran and Venugopalan (1991)

Table 2-1. Summary of Aquatic Bioconcentration Studies with Endosulfan

Chemical (formulation/ % ai) [*1]	Species	Study Design [*2]	Exposure Duration (Exposure Conc. µg/L)	BCF Method (SS) [*3]	Avg. BCF/ (BAF)	Range [SD] BCF/ (BAF)	N	Reference
	Catfish *(Mystus gulio)*	FT / M / Muscle	10-d (3 levels, 0.2- 1.95)	Ratio (SS NR)	17.1	16.6-17.5	3	
	Oyster *(Crassostrea madrasensis)*	FT / M / Foot	10-d (3 levels, 0.14- 1.41)	Ratio (SS NR)	60	42-70	3	
	Clam *(Katelysia opima)*	FT / M / Foot	10-d (3 levels, 0.14- 1.41)	Ratio (SS NR)	46	30-61	3	
Endosulfan (NR)	Crayfish *(Procambarus clarkii)*	NR / U / WB	56-d (100)	Ratio, , α+ β+ sulfate (non-SS)		≤ 1.9[*7]	---	Naqvi and Newton (1990)

[*1] TG = technical grade; ai = active ingredient; NR = not reported.
[*2] FT = flow through; R = static renewal; S = static; M = measured exposure conc.; U = unmeasured exposure conc. WB = whole body.
[*3] Ratio method = ratio of tissue to water concentration; Kinetic method = ratio of uptake to elimination rate; SS = steady state. All BCFs are expressed on a wet weight basis.
[*4] Average BCFs reported here are calculated from 9 acceptable tests reported by the authors and from treatments with no statistically significant effects on survival or growth relative to controls.
[*5] Kinetic-based BCF is questionable because elimination half-life derived from K2 is not consistent with observed data. A 21-d BCF (ratio method) of 5670 is calculated based on total endosulfan (α, β, sulfate).
[*6] BCF data included in EPA's 2002 Ecological Risk Assessment.
[*7] BCF value from this study is highly suspect due to irregular accumulation patterns and study design problems.

Table 2-2. Summary of Aquatic Bioaccumulation Studies with Endosulfan

Species	Study Location/ Design	Analytes	Water Conc. (µg/L)	Sediment Conc. (µg/kg)	Tissue Conc. (ug/kg w.w)	BAF [BSAF]	N	Reference
Mussel *(Mytilus galloprovincialis)*	Black Sea (4 coastal stations)	Endosulfan sulfate	<0.01	< 0.01-25	<0.01-0.08	[0.059]	4	Ozkoc and Bakan, 2007
Oyster *(Crassostrea virginica)*	Mesocosm (96-h, 70:30 α:β)	Total endosulfan (α+β+sulfate)	3 levels; 0.18➔0.06 0.52➔0.12 3.0➔0.29	ND (< 32)	35-606	637 ± 189	3	Pennington et al (2004)
Green alga *(Pseudokirch-neriella subcapitatum)*	Microcosm (24-h TG 2:1 α:β)	Total endosulfan (α+β+sulfate)	100	NA	53.6 (*1)	536(*1)	---	DeLorenzo et al (2002)
Water flea *(Daphnia magna)*	Microcosm (24-h TG 2:1 α:β)	Total endosulfan (α+β+sulfate)	100 (*2) 100(*2)+food food only	NA	65.6(*1) 62.4(*1) 1.68(*1)	656(*1) 624(*1) 16.8(*1)	---	DeLorenzo et al (2002)

(*1) Tissue concentrations and BCF converted from dry wt to wet wt. assuming 80% water fraction in tissue.

(*2) Water concentrations based on nominal values.

2.2 Findings From Bioaccumulation Modeling

2.2.1 Bioaccumulation in Aquatic Organisms

A preliminary application of an aquatic food web bioaccumulation model (Arnot and Gobas, 2004) was used to explore several assessment questions related to the bioaccumulation of endosulfan by aquatic organisms. This model and its precursor, (Gobas 1993) have been used extensively by USEPA for assessing bioaccumulation in the development of water quality criteria (USEPA, 1995; 2000, 2003). The primary assessment questions of interest include:

- To what extent do food web models predict bioaccumulation of endosulfan by aquatic organisms and how do these compare to measured data?
- What is the relative contribution of diet and water uptake routes to predicted concentrations in biota?
- Are piscivorous wildlife potentially at risk from predicted endosulfan concentrations in aquatic biota?

Model Inputs and Assumptions

Detailed information on all input parameters, model equations and assumptions are presented in Attachment B. Only a brief summary of input parameters and assumptions is provided below.

- **Food Web Structure:** A simple aquatic food web was assumed consisting of phytoplankton, zooplankton, filter feeding invertebrates, benthic feeding invertebrates, small and medium-size forage fish and piscivorous fish. Feeding preferences are defined in Table 9 of Attachment B and basically consist of higher trophic level organisms consuming various fractions of organisms at lower trophic levels based on typical feeding ecology for organism groups.
- **Exposure Concentrations.** Endosulfan concentrations in water were assumed to range from 0.1-5 ppb (total chemical) based on 60-d average concentrations predicted from PRZM/EXAMS for different crop exposure scenarios (USEPA, 2002). Freely dissolved concentrations in pore water were assumed equivalent to overlying water which is supported by subsequent PRZM/EXAMS modeling of pore water concentrations shown in Table 4-2.
- **Chemical Properties.** The log K_{OW} of endosulfan was assumed to range between 3.55 and 4.78 based on reported data for α and β-endosulfan (Table 11 of Attachment B). A mean K_{OC} of 13600 was used (range: 10000-16000) based on measured data (Table 10 of Attachment B). Chemical metabolic rate by biota was set to zero. Although endosulfan can be metabolized to endosulfan sulfate by biota, available data indicates this degradate is approximately equal in toxicity to the parent compounds (α and β endosulfan). Thus, the assumption of chemical metabolic rate of zero is considered reasonable.
- **Organism Characteristics.** Lipid fraction of organisms ranged from a mean of 0.5% for phytoplankton to a mean of 6% in piscivorous fish (Table 10, Attachment B). All organism physiological parameters were used as defined by Arnot and Gobas (2004).
- **Ecosystem Characteristics.** Ranges of values assumed for total organic carbon in sediment and water, suspended solids concentrations, oxygen saturation and temperature were based on information from NAWQA as shown in Table 10 of Attachment B.

The Arnot and Gobas model was run using a Microsoft® Excel spreadsheet and Monte Carlo simulations (10,000 trials of randomly selected parameters) using Crystal Ball 2000. Assumptions regarding distribution types and variance parameters are provided in Attachment B.

Model Output: Biomagnification

Detailed information on all model outputs (including a sensitivity analysis) is provided in Tables 13-15 of Attachment B. Results from mean predictions of endosulfan concentrations in aquatic organism tissues are shown in Table 2-3. Results indicate that mean predicted concentrations in tissues range form about 1.3 ppm in phytoplankton to

4.7 ppm in top piscivorous fish (wet weight basis). To evaluate biomagnification, however, tissue concentrations must be converted to a lipid weight basis. When this is done, it appears that biomagnification of endosulfan is not significant, as the calculated biomagnification factors (BMF) are near or below unity. Predicted BMF values near or below unity also occur when comparing lipid-normalized concentrations in tissue determined at higher percentiles of the distribution (e.g., 75th and 90th percentiles, lipid-normalized data not shown).

Table 2-3. Mean Predicted Concentrations and Biomagnification Factors (BMF) of Endosulfan in Aquatic Organisms at Different Trophic Levels.

Taxonomic Group	Mean Lipid Fraction	Mean Predicted Concentration (ug/kg w.w.)	Mean Predicted Concentration (ug/kg-lipid)	Mean Predicted BMF (lipid basis)
Phytoplankton	0.005	1279	255800	---
Zooplankton	0.02	1280	64000	0.25
Benthic feeding inverts	0.02	1282	64100	0.65
Filter feeding inverts	0.02	1411	70550	0.51
Small forage fish	0.06	3346	55767	0.84
Medium forage fish	0.06	3447	57450	0.87
Piscivorous fish	0.06	4682	78033	1.38

Details on model inputs, assumptions and outputs are provided in Attachment B.
BMF values calculated as the lipid-normalized concentrations in the predator divided by lipid-normalized concentrations in its diet, weighted according to feeding preferences in Table 9 of Attachment B. Lipid-equivalent concentrations in sediments determined by normalizing to sediment OC fraction * 0.35 per Seth et al. (1999) assuming negligible lipids in sediments.

Model Output: Bioconcentration and Bioaccumulation Factors

Mean bioconcentration and bioaccumulation factors predicted from the model simulations are shown in Table 2-4. Bioconcentration factors were estimated by considering endosulfan uptake through respiratory processes only while bioaccumulation factors considered both respiratory and dietary pathways. Again, the similarity in predicted BCF and BAF values indicates that the contribution of the diet to chemical accumulation is minimal, which is consistent with the moderate hydrophobicity of endosulfan. For fish, predicted BCF values range from about 1000 (mean, wet weight basis) to about 2400 (90th percentile, wet weight basis). These predicted BCF values are consistent with those observed from empirical studies reviewed in Section 2.1 (1000 to 3000 based on the highest quality studies).

Table 2-4. Mean Predicted Bioconcentration Factors and Bioaccumulation Factors of Endosulfan in Aquatic Organisms at Different Trophic Levels.

Taxonomic Group	Mean Lipid Fraction	Mean Predicted BCF (L/kg w.w.)	Mean Predicted BAF (L/kg w.w.)	90th Percentile Predicted BCF (L/kg w.w.)	90th Percentile Predicted BAF (L/kg w.w.)
Phytoplankton	0.005	499	499	1,079	1,079
Zooplankton	0.02	496	500	1,077	1,089
Benthic feeding inverts	0.02	525	530	1,122	1,132
Filter feeding inverts	0.02	515	585	1,102	1,239
Small forage fish	0.06	1,196	1,308	2,553	2,885
Medium forage fish	0.06	1,184	1,353	2,527	3,049
Piscivorous fish	0.06	1,127	1,806	2,365	4,282

Details on model inputs, assumptions and outputs are provided in Attachment B.

2.2.2 Bioaccumulation in Terrestrial Organisms

A number of recent studies have been conducted for evaluating and predicting the bioaccumulation of organic chemicals in food webs that involve terrestrial (air-respiring) organisms (e.g., Armitage and Gobas, 2007; Kelly et al., 2007; Czub and McLachlan, 2004; Kelly and Gobas, 2003; Sharp and Mackay, 2000; McLachlan, 1996). A key component from these studies has been the relationship between observed or predicted bioaccumulation of poorly metabolized chemicals in terrestrial animals and the octanol-air partition coefficient (K_{OA}). For terrestrial organisms, the relationship between the K_{OA} and bioaccumulation is somewhat analogous to the use of the octanol-water partition coefficient (K_{OW}) for predicting the bioaccumulation of nonionic organic chemicals by aquatic (water-respiring) organisms. Based on model simulations and supporting observations of chemical accumulation in Arctic terrestrial, purely aquatic and marine mammalian food webs, Kelly et al. (2007) indicate that slowly-metabolized organic chemicals with relatively low to moderate K_{OW} values (i.e., log K_{OW} between 2-5) and high K_{OA} values (i.e., log $K_{OA} \geq 6$) have the potential to biomagnify in terrestrial and marine mammal food webs but not purely aquatic food webs. In their model, the conceptual basis for biomagnification of this group of compounds by terrestrial organisms relates largely to the greater ability of terrestrial organisms to assimilate food from their diet and their slower ability to eliminate these chemicals through respiratory processes relative to aquatic organisms.

Based on a log K_{OW} of 3.7 and log K_{OA} of 7.9 for β-endosulfan, Kelly et al. (2007) calculate that biomagnification factors (BMFs) range from 2.5 to 28 for various herbivorous and carnivorous terrestrial organisms, but are all less than 1 for aquatic organisms. Importantly, these calculated BMF values assume that the chemical is not metabolized in tissues of biota. Model predictions were evaluated against measured concentrations in tissues of organisms occupying terrestrial, piscivorous, and marine arctic food webs (e.g., lichen, caribou, wolf food chain; macroalgae, zooplankton, bivalve, fish, whale/seal, polar bear food web) for a variety of chlorinated biphenyls, chlorinated benzenes, hexachlorocyclohexanes, cyclodienes (including endosulfan), and DDT-related compounds. Agreement between the mean model-predicted and observed

concentrations in biota (lipid normalized) was generally good (i.e., many fell within a factor of three and all fell within a factor of 10 of observed data).

Measured data for endosulfans reported by Kelly et al (2007) are summarized in Table 2-8. Use of these data to evaluate endosulfan biomagnification is limited because measurements only span about 1.5 trophic levels for β-endosulfan (the most complete analyte data set; TL range from 2.9 for cod to 4.5 for ringed seal). The data for lichen are germane to a terrestrial arctic food web (e.g., lichen-caribou-wolf), but endosulfan data for other components of the arctic terrestrial food web were not reported. Mean concentrations of β-endosulfan in cod (TL 2.9) and salmon (TL 3.9) are 2.9 and 0.85 ng/g-lipid equivalent, respectively, and do not suggest biomagnification is occurring in aquatic-respiring organisms just as the food web model predicts. Mean concentrations of β-endosulfan in air-respiring marine mammals are 12.6 and 4.9 ng/g-lipid equivalent in blubber of male and female beluga (TL 4.1) and 3.0 and 2.3 ng/g-lipid equivalent in blubber of male and female ringed seals (TL-4.5). Compared to concentrations in fish, these data suggest some increase in lipid-normalized concentrations in beluga, which reportedly consume invertebrates and fish. If one assumes that the fish portion of the beluga diet consists of 50% cod and 50% salmon and invertebrates have similar concentrations as fish (i.e., no aquatic biomagnification), then BMFs would be about 7 for male beluga and about 3 for female beluga. For ringed seal, which reportedly consume almost exclusively fish, a similarly calculated BMF based on 50% cod and 50% salmon would be in the 1 to 2 range for female and males, respectively. Given the limited amount of endosulfan data available from this study and uncertainties associated with the actual diets of these species, these BMF values are considered exploratory and should be viewed with caution.

Table 2.8. Concentrations of Endosulfans Measured in Components of E. Hudson Bay Arctic Food Webs (May-Sept. 1999-2001) reported by Kelly et al. (2007).

Species [TL][(1)]	No. Samples[(2)]	Tissue	% Lipid Equivalent,[(3)] Mean, (SD)	Tissue Concentration, Geom. Mean (95% CL) in ng/g lipid equivalent		
				α-Endosulfan	β-Endosulfan	Endosulfan sulfate
Lichen *(C. rangiferina)* [TL 1.0]	11	---	2.30 (0.01)	---	0.03 (---)	---
Sediment [---]	12	---	0.06 (0.04)	---	0.33 (---)	---
Cod (*B. saida*) [TL 2.9]	12	Muscle	1.12 (0.05)		2.9 (1.3-6.7)	
Salmon (*Salmo sp.*) [TL 3.9]	7	Muscle	5.41 (0.27)	0.41 (---)	0.85 (---)	0.18 (---)

Table 2.8. Concentrations of Endosulfans Measured in Components of E. Hudson Bay Arctic Food Webs (May-Sept. 1999-2001) reported by Kelly et al. (2007).

Species [TL][(1)]	No. Samples[(2)]	Tissue	% Lipid Equivalent,[(3)] Mean, (SD)	Tissue Concentration, Geom. Mean (95% CL) in ng/g lipid equivalent		
				α-Endosulfan	β-Endosulfan	Endosulfan sulfate
Beluga *(D. leucas)* (male, age 16-35) [TL 4.1]	21	Blubber	89.4 (0.53)	---	12.6 (4.5-35.1)	0.86 (0.21-3.5)
Beluga *(D. leucas)* (female, age 5-35) [TL 4.1]	14	Blubber	89.7 (1.17)	---	4.9 (1.2-19.2)	0.58 (0.11-3.0)
Ringed seal, female (*P. hispida*) TL 4.5]	7	Blubber	71.2 (2.81)	---	3.0 (0.27-33.9)	0.19 (---)
Ringed seal, male (*P. hispida*) [TL 4.5]	7	Blubber	73.4 (4.63)	---	2.3 (0.62-8.2)	0.32 (0.10-1.0)
White winged-Scoters (*M. fusca*) [TL not reported]	5	Liver	5.65 (1.25)	---	---	0.87 (---)

(1) Measurement of endosulfan in other food web components (macroalgae, bivalves, capelin, sculpin, eider ducks) were either not reported. TL (trophic level) estimated by Kelly et al (2007) from stable isotope analyses of other similar arctic food webs.

(2) Number of samples analyzed does not appear to equal the number of analyte detections.

(3) As reported by Kelley et al. (2007), lipid-equivalent fraction of sediment was determined as: % sediment organic carbon *0.35; the lipid-equivalent fraction of biota was determined from the sum of partition coefficient-weighted fractions of lipids, proteins and carbohydrates. The lipid-equivalent fraction differed from lipid fraction only for lichens, macroalgae, and bivalves.

2.3 Conclusions from Endosulfan Bioaccumulation Assessment

This preliminary assessment of endosulfan bioaccumulation considered three lines of evidence: (1) measured bioconcentration and bioaccumulation of endosulfan in aquatic organisms, (2) modeled estimates of endosulfan bioaccumulation in aquatic organisms, and (3) modeled estimates of endosulfan bioaccumulation in terrestrial organisms. Based on these lines of evidence, the following conclusions are made regarding the bioaccumulation of endosulfan.

- Measured data on endosulfan bioconcentration and bioaccumulation are limited substantially by data quality issues and do not appear to meet OPP data quality guidelines. The measured BCF and BAF values should therefore be viewed with caution.

- Given these caveats, measured BCF values in fish based on the highest quality studies range from about 1000 to 3000 (L/kg wet wt.). Considering all studies, BCF values exceeded 5000 for one fish (yellow tetra), which approximated 5700 (based on the ratio method) and 11,600 (based on the kinetic method). Substantial uncertainties in the BCF values from the yellow tetra study are noted, however, as discussed herein.

- Based on measured data, BCFs for invertebrates are lower than fish (600 or less).

- Based on one study with daphnids, endosulfan accumulation from the diet does not appear to be significant relative to uptake from water.

- Results from bioaccumulation modeling with aquatic organisms suggests that biomagnification by aquatic organisms is not likely for endosulfan. Predicted BCF values for fish are consistent with measured data and range from approximately 1000 (mean prediction) to 2400 (90^{th} percentile).

- Bioaccumulation modeling studies published in the literature indicate that biomagnification of endosulfan by terrestrial (air-respiring) organisms is a concern, with predicted BMF values ranging from 2.5 to 28 for herbivorous and carnivorous wildlife, respectively. Measured data on endosulfan in marine mammalian food webs considered in this assessment appear limited for evaluating biomagnification, and more data specific to endosulfan should be evaluated for confirming model projections of biomagnification in terrestrial organisms.

3. NEW DATA ON ECOLOGICAL EFFECTS

Following the publication of the environmental fate and ecological risk assessment chapter for endosulfan (USEPA, 2002) and in response to a 2004 Data Call-in Notice, new registrant-submitted studies were received and reviewed by EPA. The focus of these studies was on quantifying the toxicity of endosulfan sulfate to aquatic and terrestrial organisms. These studies also consisted of endosulfan sulfate toxicity resulting from sediment exposures to aquatic organisms. Quantifying the toxicity of endosulfan sulfate is necessary for evaluating the combined risk of total endosulfan residues (defined as α + β + endosulfan sulfate). In the 2002 chapter, EPA evaluated aquatic ecological risk on the basis of α- and β-endosulfan only. Furthermore, information from the published literature on endosulfan toxicity to aquatic and terrestrial organisms was not evaluated as part of the 2002 chapter. Therefore, toxicity data from the USEPA ECOTOX database were compiled for the purpose of evaluating the level of protection afforded by the registrant-submitted toxicity data. Importantly, data from ECOTOX were not reviewed further for data quality and acceptability. The sole purpose in including the ECOTOX data was to identify which literature studies have the potential to affect ecological risk conclusions.

3.1 Aquatic Organisms: Water Column Exposure

Registrant-submitted studies on the acute and chronic water column toxicity of endosulfan sulfate are shown in Tables 3-1 and 3-2, respectively. Direct comparison of the acute toxicity of endosulfan (α+β) and endosulfan sulfate toxicity within the same species (bluegill) indicates that the sulfate degradate is about equally toxic as the parent isomers (i.e., within a factor of 2). Comparison among two decapod crustaceans (grass shrimp and mysid shrimp) indicates the acute toxicity of parent isomers and the sulfate degrade are within the same order of magnitude (1.3 and 7.9 µg/L, respectively). Regarding chronic toxicity, no comparison between parent and degradate toxicity could be made among similar species**. However, in all cases, toxicity of the sulfate degradate does not alter the toxicity values used to derive aquatic acute or chronic RQ values in the 2002 assessment.** Summaries of the new registrant-submitted endosulfan sulfate toxicity studies are provided in Attachment C.

Table 3-1. Comparison Of Acute Toxicity Of Endosulfan And Endosulfan Sulfate To Aquatic Organisms Via Water Column Exposure.

Species	Endosulfan			Endosulfan sulfate	
	96-hr LC_{50} (µg/L)	48-hr EC_{50} (µg/L)	Toxicity Category (MRID)	96-hr LC_{50} (µg/L)	Toxicity Category (MRID)
Rainbow trout *Oncorhynchus mykiss*	0.8	--	very highly toxic (136999)	---	--
Bluegill sunfish *Lepomis macrochirus*	1.7	--	very highly toxic (38806)	**3.8**	very highly toxic (46382604)
Fathead minnows *Pimephales promelas*	1.5	--	very highly toxic (Mayer & Ellersieck; 05008271)	---	---
Scud *Gammarus lacustris*	--	6	very highly toxic (40094602)	---	---
Water flea *Daphnia magna*	--	166	very highly toxic (5008271)	---	---
Striped bass *Mornone saxatillis*	0.1	--	very highly toxic (00001328)	---	---
Sheepshead minnow *Cyprinodon variegatus*	---	---	---	**3.1**	Very highly toxic (46382603)[(a)]
Eastern oyster *Crassostrea virginica*	0.45	--	very highly toxic (128688)	---	---
Grass shrimp	1.3	--	very highly toxic (40228401)	---	---
Mysid shrimp *Americamysis bahia*	---	---	---	**7.9**	Very highly toxic 4640601

Table 3-1. Comparison Of Acute Toxicity Of Endosulfan And Endosulfan Sulfate To Aquatic Organisms Via Water Column Exposure.

	Endosulfan			Endosulfan sulfate	
Species	96-hr LC_{50} (µg/L)	48-hr EC_{50} (µg/L)	Toxicity Category (MRID)	96-hr LC_{50} (µg/L)	Toxicity Category (MRID)

Toxicity values in bold represent new registrant-submitted data for endosulfan sulfate. Other toxicity values are taken from the 2002 environmental fate and ecological risk assessment (USEPA, 2002)

(a) Study classified as supplemental.

Table 3-2. Comparison Of Chronic Toxicity Of Endosulfan And Endosulfan Sulfate To Aquatic Organisms Via Water Column Exposure.

	Endosulfan			Endosulfan sulfate		
Species	NOAEC (µg/L)	LOAEC (µg/L)	Endpoint (MRID)	NOAEL (µg/L)	LOAEL (µg/L)	MRID
Rainbow trout *Oncorhynchus mykiss*	0.1[(a)]	--	---	---	---	---
Fathead minnows *Pimephales promelas*	0.2	0.4	growth, survival (05008271)	---	---	---
Scud *Gammarus lacustris*	0.07	---	---	---	---	---
Water flea *Daphnia magna*	2.0	7.0	Survival (5008271)	---	---	---
Striped bass *Mornone saxatillis*	0.01[(a)]	--	---	---	---	---
Eastern oyster *Crassostrea virginica*	0.05[(a)]	--	---	---	---	---
Mysid shrimp *Americamysis bahia*	---	---	---	**0.38**	**0.73**	Growth (males) (467816-01) [(b)]

Toxicity values in bold represent new registrant-submitted data for endosulfan sulfate. Other toxicity values are taken from the 2002 environmental fate and ecological risk assessment (USEPA, 2002)

(a) Chronic value estimated using acute-chronic ratio, as described in USEPA (2002).

(b) Study classified as supplemental.

3.2 Aquatic Organisms: Sediment Exposure

Results from registrant-submitted studies on the toxicity of endosulfan sulfate resulting from sediment exposure are summarized in Table 3-3. Data on the sediment toxicity of the parent isomers were not included in the 2002 chapter (USEPA, 2002). Summaries of

the registrant-submitted endosulfan sulfate sediment toxicity studies are provided in Attachment C.

Based on the pore water concentrations from the 10-d sediment exposures presented in Table 3-3, it appears that the freshwater midge, *Chironomus tentans,* is about a factor of 10 more sensitive to endosulfan sulfate compared to the estuarine amphipod, *Leptocheirus plumulosus*. Because the effects of endosulfan sulfate on amphipod growth were not measured in the 10-d study, *L. plumulosus* may be more sensitive to endosulfan sulfate compared to *C. tentans.* Interestingly, the sensitivity difference between these two species is not evident if results are expressed as bulk sediment concentrations. This appears to reflect differences in bioavailability and subsequent partitioning into pore water between the two experiments.

Effects from exposure of *L. plumulosus* was also evaluated during a 28-d growth and reproduction study. The NOAEC and LOAEC for reduction in amphipod growth (dry wt) are 1.58 and 4 µg/L based on pore water concentrations. Effects on amphipod reproduction were observed at the same pore water concentrations as growth effects. However, results from the reproduction portion of this test may reflect adverse effects due to the solvent control since reproduction was significantly reduced in solvent controls relative to negative controls. The NOAEC of 1.58 µg/L is selected for the basis of comparison to predicted EECs for pore water and RQ calculation.

Table 3-3. Toxicity of Endosulfan Sulfate to Aquatic Organisms Through Sediment Exposure

Test Species	Exposure Duration	Endpoint [(a)]	Pore Water (µg ai/L)	Bulk Sediment (mg ai/kg dw)	Study Classification (MRID)
		Growth:			
		EC_{50}	6.4	1.9	
Freshwater Midge,		LOAEC	3.8	1.2	Supplemental
Chironomus tentans	10-d	NOAEC	**2.7**	0.56	(463826-05)
		Survival:			
Estuarine Amphipod,		EC_{50}	74	2.3	
Leptocheirus		LOAEC	45	1.6	Supplemental
plumulosus	10-d	NOAEC	27	0.86	(463826-06)
		Growth:			
Estuarine Amphipod,		EC_{50}	>4.0	>1.2	
Leptocheirus		LOAEC	4.0	1.2	Supplemental
plumulosus	28-d	NOAEC	**1.58**	0.48	(469290-01)

(a) information for the most sensitive endpoint is presented. Values in bold are used to calculate sediment toxicity RQs.

No endosulfan sediment toxicity data was available from the 2002 environmental fate and ecological risk assessment (USEPA, 2002).

3.3 Terrestrial Organisms

Results from registrant-submitted studies on the toxicity of endosulfan sulfate to avian species are summarized in Table 3-4. Based on the comparison of parent endosulfan and

sulfate degradate acute dietary toxicity with bobwhite quail and mallard duck, endosulfan sulfate appears about equal in toxicity to waterfowl (mallard) and at least a factor of 4 less toxic to game birds (quail). **The new data on endosulfan sulfate toxicity to bobwhite quail and mallard duck do not change the effect concentrations and subsequent RQ calculations presented in the 2002 ecological risk assessment.** Summaries of the registrant-submitted endosulfan sulfate sediment toxicity studies are provided in Attachment C.

Table 3-4. Comparison of Acute Toxicity of Endosulfan and Endosulfan Sulfate To Terrestrial Organisms.

	Acute Oral Toxicity				Acute Dietary Toxicity			
	Endosulfan		Endosulfan Sulfate		Endosulfan		Endosulfan Sulfate	
Species	LD_{50} (ppm)	Toxicity Category (MRID)	LD_{50} (ppm)	Toxicity Category (MRID)	5-day LC_{50} (ppm)	Toxicity Category (MRID)	5-day LC_{50} (ppm)	Toxicity Category (MRID)
Northern bobwhite quail *Colinus virginianus*	--	--	**44**	highly toxic (464305-01)	805	moderately toxic (22923)	**>3528**	(464305-02)[(a)]
Mallard duck *Anas platyrhynchos*	28	highly toxic (136998)	---	---	1053	slightly toxic (22923)	**1642**	(463826-01)
Honey bee *Apis meliferus*	4.5	– (0001999)	---	---	---	---	---	---
Laboratory rat *Rattus norvegicus*	10	highly toxic (0038307)	--	--	---	---	---	---

Toxicity values in bold represent new registrant-submitted data for endosulfan sulfate. Other toxicity values are taken from the 2002 environmental fate and ecological risk assessment chapter (USEPA, 2002).

(a) Study classified as supplemental.

4. NEW INFORMATION ON AQUATIC EXPOSURE

The previous ecological risk assessment for endosulfan (USEPA, 2002) contained an aquatic exposure estimates (including drinking water) as well as an ecological risk assessment; risk quotients (RQs) were calculated based on total (α- and β-) endosulfan. EFED has subsequently revised exposure modeling to include the racemic mixture of parent compounds and their primary degradate, endosulfan sulfate all of which have similar toxicities. PRZM and EXAMS are screening simulation models coupled with the input shell pe5v01.pl (Aug.8, 2007) to generate daily exposures and 1-in-10 year estimated environmental concentrations (EECs) of total endosulfan residues that may occur in surface water bodies adjacent to application sites.

The appropriate chemical-specific PRZM input parameters were selected from reviewed environmental fate data submitted by the registrant and in accordance with EFED water

model input parameter selection guidance. Some of the input parameters are similar to those used in the 2002 science chapter (USEPA, 2002); no new environmental fate data were incorporated into this assessment. However, upper bound input parameters were used in absence of a complete suite of environmental fate data for total toxic residues of endosulfan. A summary of the chemical-specific model inputs used in this assessment are provided in Table 4-1. PRZM/EXAMS model EECs representing 1-in-10 year peak, 21-day average, and 60-day average concentrations of total endosulfan residues in surface water and benthic sediment pore water are presented in Table 4-2 and full set of EECs are given in Attachment D. Tomatoes and strawberries were modeled since they represent major uses of endosulfan and these uses rely on some of the highest registered use rates. The majority of tomato and strawberry production practice in Florida and California use plastic mulch. Recent studies show significantly greater loss of endosulfan due to larger volumes of runoff water from plastic mulch (compared to vegetative mulch) resulting in increased loading of both dissolved and particle-bound endosulfan residue (McCall *et al*. 1998; Rice 2001). Although plastic mulch is typically used for both of these crops, PRZM/EXAMS scenarios are not currently available to model the effects of plastic mulch on runoff; as a result, surface water estimates may underestimate surface water concentrations in waterbodies adjacent to these modeled uses.

Table 4-1. PRZM/EXAMS environmental fate input parameters for total endosulfan

chemical	Endosulfan	Source	Commnets[‡]
Molecular weight	406.9	Product Chemistry	α and β endosulfan
Solubility[†]	530 µg/L	MRID 404215-02	α endosulfan
Vapor pressure	7.2x 10^{-7} torr	MRID 414215-01	β endosulfan
pH 7 hydrolysis half life	19 days	MRID 414129-01	α endosulfan
Aqueous photolysis half life (near surface)	stable	MRID 404215-02	α and β endosulfan
Soil photolysis half life	stable	MRID 414307-01	α and β endosulfan
Aerobic soil metabolism half life	1335.6 days (upper 90% c.i.)	MRID 438128-01	Total endosulfan
Aerobic aquatic metabolism half life	2671.2 days (2 x 1335.6 days soil metabolism PRZM/EXAMS value)	EFED Guidelines	--
anaerobic aquatic metabolism half life	382. days (2 x 196 upper 90% c.i. of anaerobic soil study)	MRID 414129-04	β endosulfan
Soil organic carbon partitioning (Koc)	10600 L kg^{-1} (mean value PRZM/EXAMS)	MRID 414129-06	α endosulfan

Table 4-1. PRZM/EXAMS environmental fate input parameters for total endosulfan

chemical	Endosulfan	Source	Commnets‡
Crop	FL Tomato and CA Strawberry		---
application rate	3.0 lbs a.i. acre	Product label	---
Number of applications	3	Product label	---
Application method	aerial	Product label	---
Application dates	September 15 and June 15		---
spray efficiency	95%	EFED Guidelines†	--
spray drift	5%	EFED Guidelines†	---

† = Water solubility was multiplied by 10 according to Guidance for selecting input parameters in modeling for environmental fate and transport of pesticides Version II. February 27, 2002.
‡ In absence of fate date for total endosulfan, conservative input fate parameters were selected for PRZM/EXAMS modeling

Table 4-2: EEC's of total endosulfan residues (α+β+ endosulfan sulfate) for surface and benthic sediment pore water for Florida tomatoes scenarios

Medium	Acute: Peak EEC (µg/L)	Chronic: 21-day Average EEC (µg/L)	Chronic: 60-day Average EEC (µg/L)
Florida Tomatoes and 3.0 lbs a.i./ acre			
Surface Water	23	9.3	6.8
Pore water	4.5	4.4	4.3
California Strawberry and 3.0 lbs a.i./acre			
Surface Water	12	5.5	3.9
Pore water	2.5	2.5	2.4

5. NEW INFORMATION ON MONITORING AND LONG-RANGE TRANSPORT

Endosulfan is a semi-volatile and persistent compound belonging to the cyclodiene class of pesticides. Endosulfan consists of two enantiomers (α and β endosulfan and the technical grade pesticide is typically a racemic mixture of the two enantiomers in the ratio 30:70 (alpha:beta). The chemical can migrate over a long distances through various environmental media such as air, water, and sediment. Once endosulfan is applied to crops, it can either persist in soil as a sorbed-phase or dissipate from the site of application through several physical, chemical, and biological processes.

The occurrence of endosulfan in remote regions like the Great Lakes, Arctic, and mountainous areas are well documented. Recent studies suggest that desorbed residues

of endosulfan volatilize and continue to recycle in the global system through a process of migration and redeposited via wet and dry depositions as well as air-water exchange in the northern Hemisphere. Dust dispersion and translocation also contribute endosulfan into the atmosphere as adsorbed phase onto suspended particulate matter, but this process does not appear to be a major contributor like volatilization. Transport of endosulfan in solution and sediment bound residues also can potentially contribute in the long-range and regional distributions of endosulfan.

Monitoring data indicate that endosulfan is moving through various environmental media such as air, water, and sediment. However, these data likely under represent actual field residues, since monitoring efforts are mostly non-targeted. Data from non-targeted monitoring also pose uncertainties in spatial and temporal distributions of endosulfan residues in relation to endosulfan use. Because of the persistence of endosulfan and it's degradate in the environment, these compounds can travel long distances. Evidence for regional and long- range transport of endosulfan is provided below from a large number of literature sources reporting concentrations in various environmental media.

Endosulfan in surface water

The presence of endosulfan in many ecosystems throughout the U.S. has led to concern regarding continuing point and non-point sources of endosulfan in vulnerable areas that may result in acute and chronic effects on aquatic communities. Currently, endosulfan is one of the major insecticides used in vegetable production in southern Florida. Since 1991, the South Florida Water Management District's (SFWMD) non-target quarterly water quality monitoring program has been analyzing a number of pesticides including endosulfan at 34 sites (Figure 5-1). Endosulfan and endosulfan sulfate were detected in surface waters and benthic sediments at several locations in the south Miami-Dade County farming area. Endosulfan has been measured at concentrations exceeding the chronic surface water quality standard of 0.056 μg/L (Figure 5-2) for a number of years (assuming endosulfan sulfate has similar toxicity to parent endosulfan).

The University of South Carolina (USC) and the National Oceanic and Atmospheric Administration (NOAA) also conducted a monitoring study targeting areas where endosulfan was used (Delorenzo *et al.*, 2001). The USC/NOAA monitoring data have been compared with data collected by the SFWMD; total toxic residues of endosulfan (alpha and beta endosulfan plus endosulfan sulfate) in both studies collected at similar locations and times are roughly equivalent (Table 5-1). These data suggest that in the vicinity of row crops where endosulfan is reportedly applied, endosulfan residues have been routinely detected in both the water column and benthic sediments. Additionally, the data indicate that total endosulfan residues have moved to areas distant from where it was initially applied and that the residues are sufficiently high, when compared to toxicity values of aquatic organisms to exceed the Office of Pesticide Programs' (OPP) acute and chronic risk levels of concern.

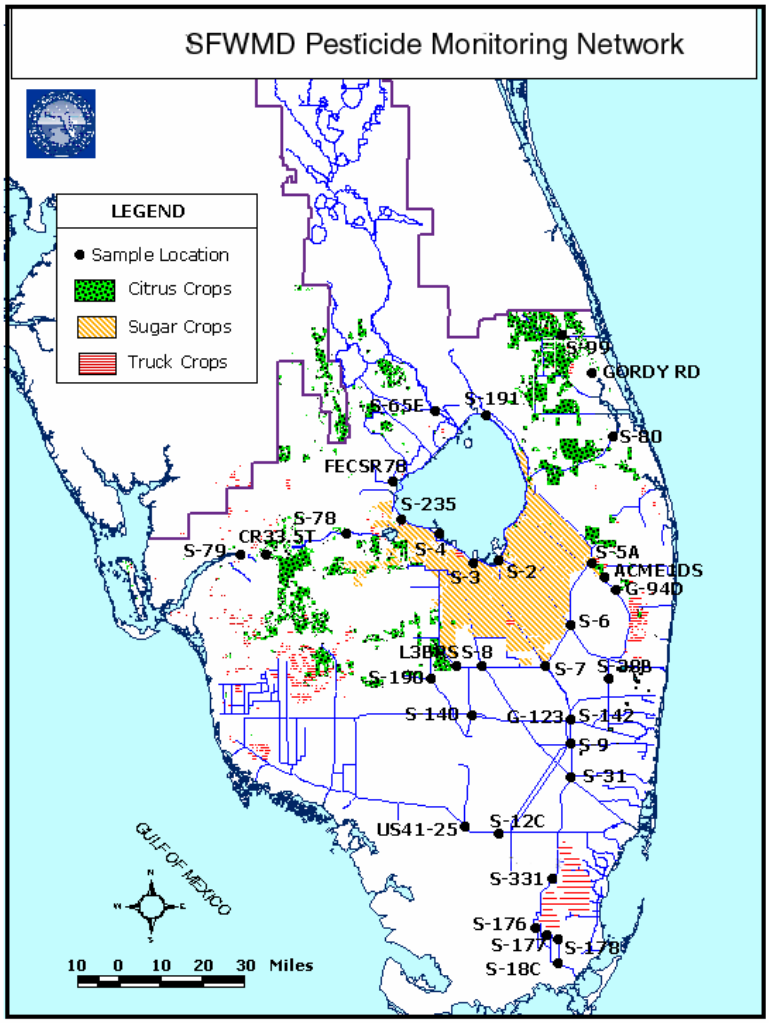

Figure 5-1. South Florida Water Management District's surface water and sediment collection sites in South Florida (Pfeuffer and Matson. 1998 to 2007).

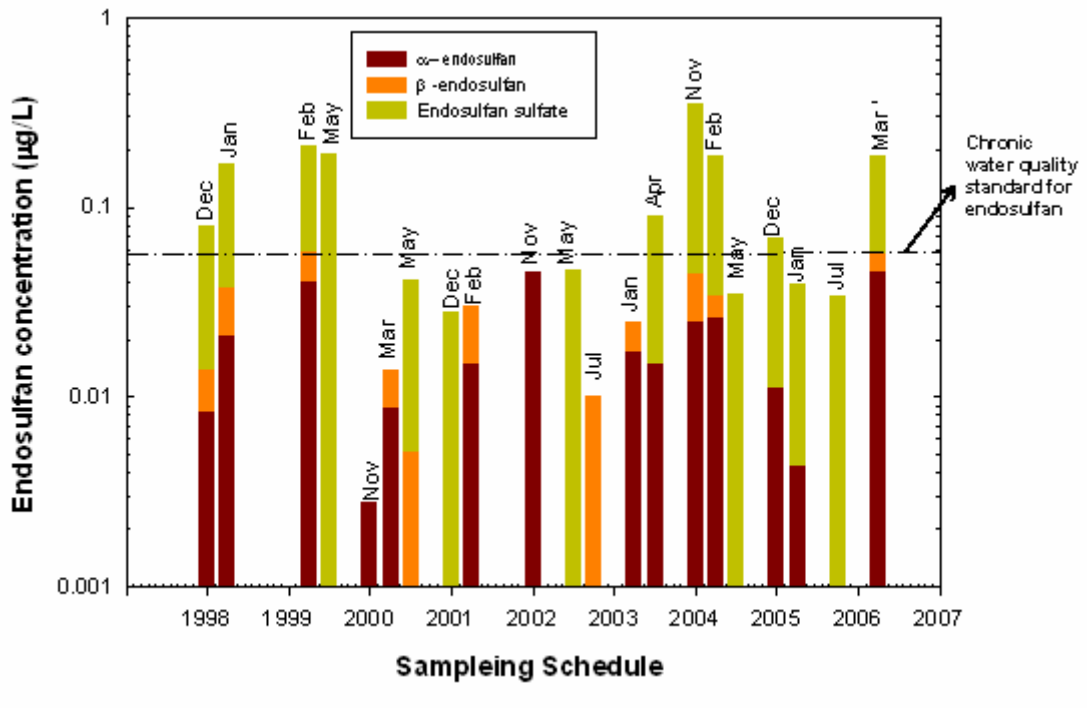

Figure 5-2. Concentrations of parent and endosulfan sulfate in surface water samples from site S178, South Florida

Results of the field studies conducted during 2002 -2004 by Herman-Fetcho et al. (2005) and 1993 -1997 by Scott *et al*. (2002) also indicate the presence of endosulfan in surface water samples from southern Florida and Florida Bay. In a two year study, endosulfan was frequently detected in the South Florida canals and Biscayne Bay, with an average concentration of 11 ng/L (Herman-Fetcho *et al*., 2005). Endosulfan concentrations were higher near vegetable production areas where endosulfan is applied. The study also indicates that endosulfan has the highest hazard potential to aquatic organisms among the pesticides evaluated. Scott et al. (2002) reported that endosulfan was detected at 100% of the sites sampled. Endosulfan residues in surface waters from irrigation canals and Florida Bay occasionally exceeded the chronic water quality criterion. While endosulfan concentrations at these sites did not result in any reported fish kills, residues exceeded concentrations that are known to cause chronic effects in copepods, clams, and oysters.

Table 5-1. Summary of South Florida Water Management District (SFWMD) and National Oceanic and Atmospheric Administration/ University of South Carolina (NOAA/USC) monitoring data collected at Site S178 and the C-111 Canal, Dade County, Florida.

Site 178 (SFWMD)				Site C (NOAA/USC)			
Date	**Total Endosulfan µg/L**	**Fish Chronic RQs**	**Invertebrate Chronic RQs**	**Date**	**Total Endosulfan µg/L**	**Fish Chronic RQs**	**Invertebrate Chronic RQs**
Aug 98				Aug 10, 98 Aug 13, 98 Aug 14, 98	0.002 0.010 0.008		
Dec 98	**0.066**‡	**0.6**	**0.94**	Oct 27, 98			
Jan 99	**0.167**‡	**1.51***	**2.39****	Feb 8, 99 Feb 11, 99 Feb 12, 99	**0.173**‡ **0.186**‡ **0.151**‡	**1.57*** **1.69*** **1.37***	**2.47**** **2.66**** **2.16****
Apr 99				June 4 99 Jun 11, 99	**0.067**‡ 0.041	0.61 0.37	**0.96** **0.57**
Aug 99				No Sampling			
Nov 99	ND			Oct 1, 99 Oct 3, 99 Oct 5, 99 Oct 7, 99 Oct 8, 99	0.0012 0.0012 0.0053 0.0045 0.0058		
Feb 00	**0.208**‡	**1.89***	**2.97****	Feb 10, 00 Feb 12, 00 Feb 14, 00 Feb 16, 00 Feb 17, 00	**0.210**‡ **0.187**‡ **0.183**‡ **1.345**‡ **0.256**‡	**1.91*** **1.70*** **1.66*** **12.22*** **2.33***	**3.00**** **2.67**** **2.61**** **19.21**** **3.66****

Table 5-1. Summary of South Florida Water Management District (SFWMD) and National Oceanic and Atmospheric Administration/ University of South Carolina (NOAA/USC) monitoring data collected at Site S178 and the C-111 Canal, Dade County, Florida.

Site 178 (SFWMD)				Site C (NOAA/USC)			
May 00	**0.19**‡	**1.72***	**2.71****	June 6, 00 June 8, 00 June10, 00	**0.056**‡ 0.051 **0.102**‡	**0.51** 0.46 **0.92**	**0.80** **0.73** **1.46****
Aug 00				Sep 25, 00 Sep 27, 00 Sep 29, 00	0.002 0.015 0.013		

* Exceeds chronic freshwater fish Level of Concern (RQ ≥ 1.0) based on an NOEC = 0.11 µg/L.

** Exceeds chronic freshwater invertebrate Level of Concern (RQ ≥ 1.0) based on an NOEC of 0.07µg/L.

‡ Exceeds Florida Class III water quality standard of 0.056 µg/L.

California Department of Pesticide Regulation, Environmental Hazard Assessment Program (EHAP), United States Geological Survey (USGS), and the Central Valley Regional Water Quality Control Board carried out pesticide monitoring studies for surface water (CDPR 2000). Data from these and other studies are documented in EHAP's surface water database (SURF). At present, SURF contains more than 93000 pesticide analysis records for 146 chemicals. Data summarizes for each pesticide include the number of analyses, frequency of detection, the 95^{th}, 75^{th}, and 50^{th} percentile concentration for the period of 1990 to 1998. Endosulfan sulfate had the highest detection frequency at 17.2% and the 95_{th} percentile concentration was 0.14 µg/L compared to the detection frequencies of 5.2% to 5.4% and the 95_{th} concentrations of 0.11 and 0.07 µg/L for parent endosulfan and β-endosulfan, respectively. Furthermore, in 1998, Calleguas Creek in California and the Yakima River in Washington State were classified as impaired water bodies under the Section 303d of the Clean Water Act due to presence of endosulfan as well as other chemicals.

Water samples from four temperate lakes in south-central Canada show the presence of α-and β-endosulfan (Muir *et al.*, 2004). Mean concentration levels of α-endosulfan ranged from 28.5 – 1.3 pg L^{-1}, and those of β-endosulfan from 10.3 – 0.0 pg L^{-1} in lakes Opeongo, Nipigon, Britt Brook, and Virgin pond. No agricultural area was within 31 miles (50 Km) of any of these lakes, suggesting that the presence of endosulfan resulted from atmospheric transportation and deposition. Monitoring and modeling results suggest that under the conditions prevailing in south-central Canada, endosulfan can potentially undergo regional-scale atmospheric transport and reach lakes outside endosulfan use areas.

Recent monitoring data for endosulfan shows the presence of endosulfan in waters of isolated lakes in Ontario and New Brunswick (UNEP, 2002). Endosulfan, was detected in all lake trout collected from these isolated lakes; endosulfan tissue residues ranged from <0.1-0.8 ng g^{-1}ww. Endosulfan was higher in Labrador lakes. The results suggest the wide dispersal of endosulfan from areas of use to isolated lakes.

Endosulfan in atmosphere

Detailed atmospheric concentrations of α-endosulfan and β-endosulfan were summarized by Ngabe and Bidleman (2001) in North America. Early measurements of endosulfan in air were made during a survey of airborne pesticides across the United States in 1970 (Majewski and Capel,1995). Mean concentrations of α-endosulfan ranged from 0.7 ng m^{-3} in Meadow, North Carolina, to 159 ng m^{-3} in Peaksmill, Kentucky. The average concentrations of α- and β-endosulfan in air were 0.170 and 0.045 ng m^{-3} at Solomons, Maryland, in 1995 (Harman-Fetcho *et al*., 2000). The frequency of occurrence of α- and β-endosulfan in monitoring samples was 100%.

Air Resource Board (ARB) of California monitored an endosulfan application to an apple orchard in San Joaquin County in April 1997, and conducted ambient air monitoring during a period of high use of endosulfan in Fresno County in July-August 1996 (ARB, 2002). Air concentrations of α-endosulfan ranged from 3800 ng·m^{-3} and 290 ng·m^{-3}. The

detections for β-endosulfan during the same sampling period ranged from 200 $ng \cdot m^{-3}$ to 48 $ng \cdot m^{-3}$. The ratio of α-isomer: β-isomer varied from 5 to 209 across all the samples with concentrations of both isomers above the limit of quantification (LOQ).

Some monitoring in California for endosulfan coincided with expected applications to grapes and cotton. The maximum concentrations in ambient air were 140 $ng \cdot m^{-3}$ for α-endosulfan and 26 $ng \cdot m^{-3}$ for β-endosulfan. The highest average concentrations for various sites were 24 $ng \cdot m^{-3}$ for α-endosulfan and 5.4 $ng \cdot m^{-3}$ for β-endosulfan. All the highest concentrations occurred at one site in the town of San Joaquin, CA, which is three quarters to one mile from the closest endosulfan use area.

Abundant regional concentration data are available for the Great Lakes Region from a joint US EPA / Environment Canada-monitoring project IADN (Integrated Atmospheric Deposition Network) (Sun *et al*., 2006) and Sun *et al.* (2003) providing compelling evidence for medium-range airborne transport of endosulfan and endosulfan sulfate. The endosulfan concentrations (shown as the sum of α- and β-endosulfan) in vapor phase showed a clear increasing trend from the west to east (Figure 7), except for the remote site of Burnt Island. At each site, the average concentration was skewed by high outliers that usually occurred in the summer and were attributed to current agricultural use of endosulfan. Higher endosulfan concentrations were observed at Point Petre, Sturgeon Point, and Sleeping Bear in vapor, particle, and precipitation phases, which could be explained by its heavy usage in the surrounding areas (Hoh and Hites, 2004). For example, endosulfan is widely used in Michigan and New York State (Hafner and Hites, 2003) and in Ontario (Harris, et al., 2001), particularly in the southern and western portions of the province.

Total endosulfan concentrations showed no long-term decreasing trends in the vapor phase at Eagle Harbor (EH), Sleeping Bear Dunes (SBD), or Sturgeon Point (SP) (Figure 5-3). However, total endosulfan concentrations in the particle phase declined at all five U.S. sites. In the precipitation phase, total endosulfan concentrations only decreased at Point Petre (PP), while at the other six sites, these concentrations did not change from 1997 to 2003. The National Center for Food and Agriculture Policy provides an endosulfan usage database for the period 1992-97 in the U.S. Although endosulfan usage in Michigan significantly decreased from 29 tons to 19 tons between 1992 and 1997, increasing usage was also observed in the surrounding states, including New York, Indiana, Kentucky, and Minnesota. Because of the lack of updated usage data, correlation between the decreasing particle-bound endosulfan concentrations and its usage pattern is difficult.

Total endosulfan concentrations also showed a strong seasonal variation in precipitation. The ratio between the highest and the lowest total endosulfan concentration ranged between about 2-10. In particular, this ratio is as high as 10 at Point Petre, suggesting a heavy usage in the surrounding area. At all sites, the total endosulfan concentrations peaked in early July in precipitation, a time which corresponds well with its maximum agricultural usage.

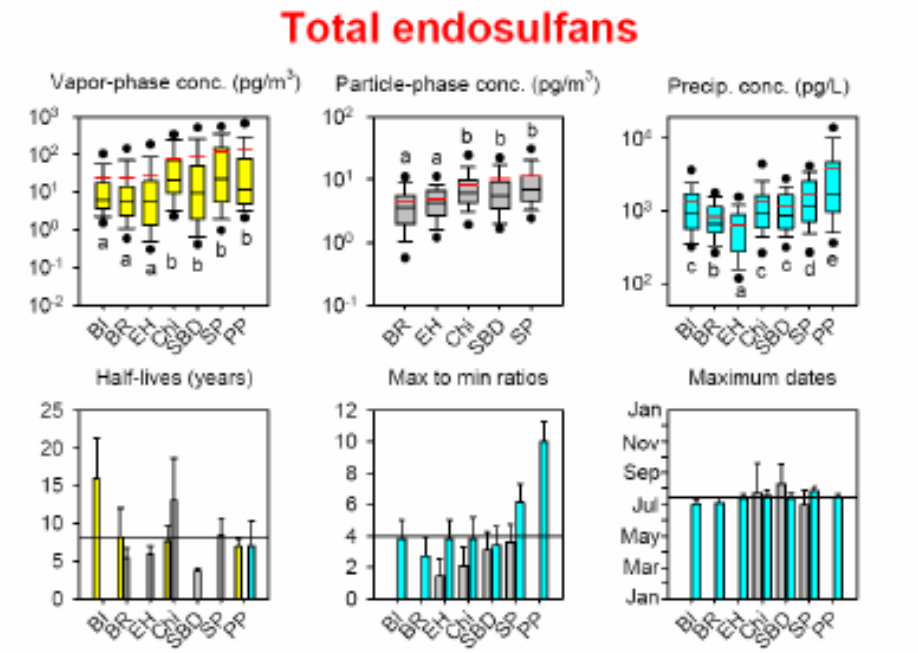

Figure 5-3. Spatial and temporal trends of total endosulfans (sum of α- and β-endosulfan).

Shen *et al*. (2006) evaluated endosulfan concentration in air using passive air samplers (PAS) to trap endosulfan. Gaseous concentrations of endosulfan varied from 3.1 to 681 pg·m^{-3} for α-endosulfan and from 0.03 to119 pg·m^{-3} of β-endosulfan. The maximum measured concentration of endosulfan in air was generally lower than 58 pg·m^{-3} across North America. The highest measured concentrations were reported in the Okanagan Valley, British Columbia, East Point on Prince Edward Island, Manitoba, and Tapachula, Mexico

Endosulfan in Precipitation

Several studies demonstrated that endosulfan is removed from the atmosphere by rain and snow fall. In a monitoring study carried out in eastern Canada between 1980 and 1989, α-endosulfan was reported occasionally at concentrations near the detection limit of 10 ng L^{-1} (Brun et al. (1991). In precipitation of the Great Lakes region, α- and β-endosulfan concentrations were regularly determined by IADN at various stations during the period of 1987–1997. Concentration levels of α-endosulfan ranged from 0.13 – 1.95 ng·L^{-1} and those of β-endosulfan from 0.19 – 6.09 ng·L^{-1} in Lake Superior and Lake Erie. Higher values were reported from Lake Michigan ranging from 0.54 – 8.22 ng·L^{-1} for α- and from 1.06 – 12.13 ng·L^{-1} for β-endosulfan [36]. Unlike for vapor- phase concentrations, it has been observed that the β-isomer was often higher in precipitation than the α-isomer. This equal or greater observed wet deposition of β-endosulfan compared to α-endosulfan might be explained by the comparatively higher importance of particle vs. gas-phase scavenging. Concentrations of the transformation product endosulfan sulfate measured in precipitation of the Great Lakes region were mostly in a range of 0.1 to 1 ng·L^{-1}.

Endosulfan and endosulfan sulfate were detected in seasonal snowpack samples at six national parks in the Western United States (Hagman et al., 2006). Concentrations of total endosulfan concentrations were 1.5 $ng \cdot L^{-1}$ to 0.0040 $ng \cdot L^{-1}$ in the Sequoia, Mount Rainier, Denali, Noatak-Gates, Glacier and Rocky Mountain National Parks. The percentage contribution of endosulfan sulfate to the total endosulfan concentration ranged from 4.0% to 57.0% with mean value being 24.0%. The study results suggest that current use of endosulfan plays a significant role in contributing to the deposition of endosulfan via snow to remote high-elevation and high-latitude ecosystems.

Endosulfan on Airborne Particles

Within the IADN project, endosulfan concentrations were also measured in airborne particulate (filter-retained) matter. Average concentration levels were approximately 7.5 $pg \cdot m^{-3}$ for α-endosulfan and 2.9 $pg \cdot m^{-3}$ for β-endosulfan from 1995 to 2000. Seasonal differences for particles were much less pronounced as compared with the gas-phase data. Endosulfan associated with airborne dust was also measured on a cotton farm in Australia during the growing season. Total endosulfan residues (α- + β- + -sulfate) in airborne dust ranged from 0.07 to 1.04 $\mu g \cdot g^{-1}$ [Leys et al. (1998)].

Endosulfan in Sediment

The presence of endosulfan in the sediments is well documented in the National Sediment Contaminant Point Source Inventory (NSI) databases prepared by the Office of Science and Technology (OST) of US EPA (EPA-823-C-01-001). EPA's evaluation of the NSI data was the most geographically extensive investigation of sediment contamination ever performed in the United States. In the NSI data base, 199 detections for α-endosulfan, ranged from 0 to 11000 $\mu g \cdot Kg^{-1}$; 667 detections for β-endosulfan, ranged 0 to 67500 $\mu g \cdot Kg^{-1}$, and 195 detections for endosulfan sulfate ranged from 0.2 to 900 $\mu g \cdot Kg^{-1}$ (after culling data to eliminate dubious data, e.g. ND and < codes) in the sediments were reported between 1980 and 1999 (Figure 5-4).

Seventy sediment samples were collected over a 10-county area in the agriculture-dominated Central Valley of California, with most sampling sites located in irrigation canals and small creeks, to investigate the distribution of 26 pesticides including endosulfan (Weston *et al*., 2004). Total endosulfan concentrations in sediments ranged from 571 $\mu g \cdot Kg^{-1}$ to <1.0 $\mu g \cdot Kg^{-1}$. They also investigated the sediment toxicity of endosulfan. Measured 10-day LC_{50} values for *C. tentans* were 0.96, 3.24, and 5.22 $mg \cdot Kg^{-1}$ of organic carbon (oc) for α -, β -, and endosulfan sulfate respectively. Measured 10-day LC_{50} values for *H. azteca* were 51.7, >1000, and 873 $mg \cdot Kg^{-1}$ of organic carbon for α-, β-, and endosulfan sulfate, respectively. Endosulfan concentrations were below the acute toxicity of aquatic invertebrates in the majority of samples; however, the study suggests that endosulfan may have contributed to toxicity in the tailwater ponds or a few irrigation canals where concentrations exceeded several hundred $\mu g \cdot Kg^{-1}$. Endosulfan residues have been detected in several sites (Figure 5-4) in south Florida. The concentrations of endosulfan in sediment samples ranged from 100 $\mu g \cdot Kg^{-1}$ to non-detect.

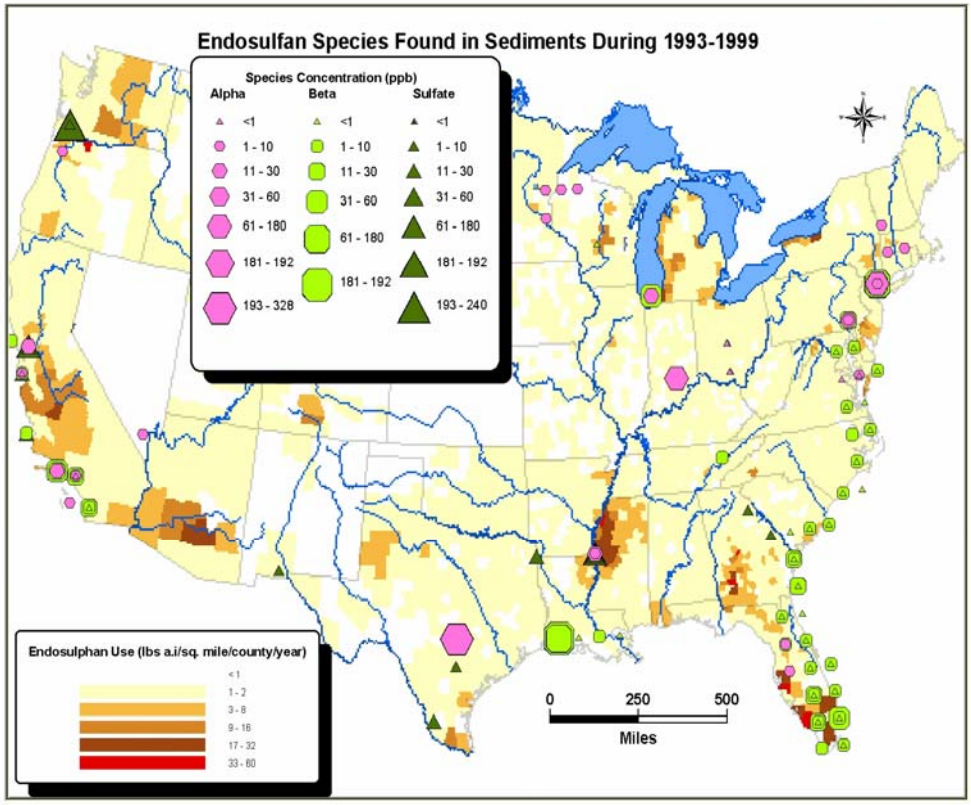

Figure 5-4. Spatial distribution of endosulfan in sediments

Endosulfan in Mountainous Regions

The effect of "global distillation" which is believed to account for transport of persistent organic pollutants (POPs) whereby a compound could volatilize from warmer regions, undergo long-range atmospheric transport and subsequently recondense to an accumulation of these substances in the temperate, higher mountainous and Arctic regions. Wania and Mackay (1993) suggested that, through "global distillation" of organic compounds could become latitudinally fractionated, "condensing" at different temperatures according to their volatility, so that compounds with vapor pressures in a certain low range might accumulate preferentially in polar regions. Endosulfan was found in the atmosphere of European mountain areas (Central Pyrennes and High Tatras). Like hexachlorocyclohexane (HCH), endosulfan was found in higher concentrations in the warm periods (4-10 pg m^{-3}) in the gas phase and particulate phase, reflecting their seasonal use pattern (Drooge and Grimalt 2004*)*. Many POP substances as well as endosulfan were found in snowpack samples collected at different altitudes of mountains in western Canadian (Blais et al., 1998). The levels of contaminants in snow and in snowpack increased with the altitude. The concentration range of α-endosulfan was 0.06–0.5 ng L^{-1} in the sampling altitude range of 700 – 3,100 m. Aerial transport also caused contamination of snow (Sequoia National Park) and water (Lake Tahoe basin) of the Sierra Nevada Mountains in California, a region adjacent to the Central Valley which is

among the heaviest pesticide use areas in the U.S.. Levels of α-endosulfan found in rain were in a range of < 0.0035 ng/L to 6.5 ng L^{-1} while β-endosulfan was determined at concentrations of < 0.012 ng L^{-1} up to 1.4 ng L^{-1} McConnell et al. (1998).

The concentrations of α-, β-endosulfan and the endosulfan sulfate were present in water from both Tablelands and Sixty Lakes in the Sierra Nevada Mountains of California (Fellers et ala., 2004). Only small differences occurred in the levels of the α- and β-endosulfan from the Tablelands and Sixty Lakes (Table) sampling areas, but the sulfate concentrations were almost an order of magnitude higher at the Tablelands compared with Sixty Lakes (Table 5-2). In frog tissue samples, only the α-endosulfan isomer was observed at levels above quantitation limits and concentrations at the Tablelands sites were not significantly different from the Sixty Lakes sites.

Table 5-2. Concentration of pesticides (ng/L) in surface water samples collected at the Tablelands (Sequoia National Park), and the Sixty Lake Basin (Kings Canyon National Park) California, USA.

Compound	Detection limit	Tablelands		Sixty Lakes	
		G049	G054	S545	S 471
α-endosulfan	0.03	0.78	1.0	0.30	0.37
β-endosulfan	0.03	0.40	0.42	1.8	0.17
Endosulfan sulfate	0.03	2.9	2.2	0.33	0.40

For mountain lakes in the Alpes, Pyrenees (Estany Redò and Caledonian Mountains (Øvre Neådalsvatn (Norway), via atmospheric deposition of endosulfan was estimated between 0.2 and 340 ng m^{-2} per month (Carrera et al., 2002). Unlike for other chemicals, endosulfan showed a more uniform geographical distribution, the lakes in the South were much more exposed to endosulfan impact, reflecting the impact of agricultural activities in southern Europe. In the northern lake only the more recalcitrant endosulfan sulfate was determined.

Endosulfan in Arctic Areas

Text in *italics* and selected figures are verbatim from the draft dossier prepared in support of a proposal of endosulfan to be considered as a candidate for inclusion in the Annexes to the Stockholm Convention prepared by the German Federal Environment Agency – Umweltbundesamt, Dessau (Feb 2007). Most of the literature cited in the Arctic for this section by GFEA (2007) related to endosulfan were originally cited in the "Endosulfan in the Atmosphere, Review and Evaluation" by Ngabe and Bidleman 2001. Recently a report (MRID 467343-01) has been submitted to the Agency by the Endosulfan Task Force (ETF) summarizing and interpreting environmental data on endosulfan in arctic regions. The primary objective of this report is to assemble broader basis for evaluation of behavior and exposure of endosulfan and characterizing uncertainties in the critical studies related to endosulfan in the Arctic. The ETF suggested that future endosulfan

monitoring can be improved through changes in analytical strategy and monitoring design to reduce uncertainties in the Arctic monitoring. However, at this time the Agency has not had sufficient time to thoroughly review the uncertainties reported by the ETF.

Various persistent chemicals contamination is a serious global threat including the Arctic. When air masses carrying contaminants reach the Arctic, the "global distillation" occurs. The air contaminants move from the gas or vapor phase into a liquid phase, and are carried to the ground in rain or snow. Once the persistence pollutants reach the Arctic, the cold temperatures and long, dark winters slow the degradation process. Polar ice can trap contaminants that are gradually released into the environment during melting periods, even years after their arrival in the Arctic. As a result, the Arctic acts as a final "sink" where pollutants from around the world accumulate and become trapped. Long range atmospheric transport of α- and β-endosulfan to the Arctic was first noticed in 1986–1987 (Patton et al. 1989). A "brown snow" event occurred in the central Canadian Arctic during the year 1988. The snow was colored by dust that appeared to be transported from western China. Endosulfan was detected in the dust at a concentration of 22 pg L^{-1}. Since then endosulfan has been routinely found in the Canadian Arctic air monitoring program, from 1993 up to the present (Halsall et al., 1998; Hung et al., 2001). Extensive monitoring data of endosulfan from the Arctic are available for the atmosphere, snowpack, surface water and biota (Bidleman et al., 1992; De Wit *et al*., 2002; Hallsall et al., 1998; Hobbs *et al*, 2003; Jantunen and Bidleman, 1998).

Air

Endosulfan was reported as a widely distributed pesticide in the atmosphere of Northern polar regions [37]. Unlike for most other organochlorine pesticides average concentrations of endosulfan in the Arctic have not changed significantly during the last five years [38][1]. Concentrations of endosulfan (isomers unspecified) from Arctic air monitoring stations increased from early to mid-1993and remained at that level through the end of 1997 at 0.0042-0.0047 ng/m3. No clear temporal trends of endosulfan concentrations in the arctic atmosphere [39][2]. Measurements taken in air at Alert, Nunavut, Canada resulted in annual average concentrations between 3 and 6 pg/m3 during 1993 to 1997. Fluctuating values mirror the seasonal applications in source regions (ref. figure 4-1)[3].

[1] *Meakin, S. What´s New with POPs Research in the Arctic. Northern Perspectives 26 (1), 6-7 (2000)*
[2] Hung, H., C.J. Halsall, P. Blanchard, H. Li, P. Fellin, G. Stern, B. Rosenberg. Temporal trends of organochlorine pesticides in the Canadian Arctic atmosphere. Environ. Sci. Technol., 36, 862-868, (2002)
[3] GFEA (German Federal Environment Agency). 2007. Draft Dossier prepared in support of a proposal of endosulfan to be considered as a candidate for inclusion in the UN-ECE LRTAP protocol on persistent organic pollutants. German Federal Environment Agency. Umweltbundesamt, Berlin. http://www.unece.org/env/popsxg/docs/2007/Dossier_Endosulfan.2007.pdf

Concentrations of endosulfan in Arctic air were found to be exceeded only by those of ΣHCH-isomers and HCB [40][1] (ref. figure 4-2)[2]. In comparison to monitored concentrations in the Great Lakes region, atmospheric levels in the Artic were less dependent on temperature, although seasonal variations were apparent as well. For example α-endosulfan concentrations ranged a factor of 3-5 over spring to fall periods. This infers a more blurred bimodal seasonal cycle with growing distance from areas of application. Hung et al. [41][3] used temperature normalization, multiple linear regression, and digital filtration to analyze the temporal trends of an atmospheric dataset on organochlorine pesticides (OCs) collected at the Canadian high arctic site of Alert, Nunavut. While air concentrations of Lindane and Chlordane showed decreasing trends through the 1990s with half-lives of 5.6 and 4.8 years α- endosulfan showed a very slow decline with a half-live of 21 years.

Seasonal variation of concentrations was also reported from Sable Island (240 km east of Nova Scotia at 43°57′N, 60°00′W). In summer aerial endosulfan concentration (α- and β-isomer) were determined between 69 and 159 ng/m3 while for wintertime values dropped to 1.4-3.0 pg/m3 (only α -isomer) [42][4].

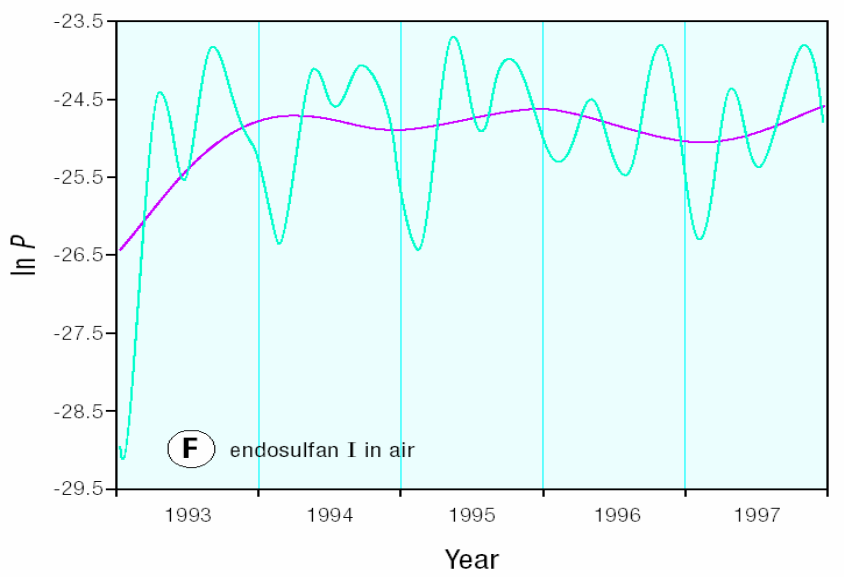

Figure 4-1 Trend (purple) and seasonal cycle (blue) of α-endosulfan in air at Alert, Nunavut, Canada (82°30′N, 62°20′W); source [39]

[1] *Halsall, C.J., R. Bailey, G.A. Stern, L.A. Barrie, P. Fellin, D.CG. Muir, B. Rosenberg, F.Ya. Rovinsky, E.Ya. Kononov, B. Pastukhov. Multi-year observations of organohalogen pesticides in the Arctic atmosphere. Environmental Pollution 102, 51-62, (1998)*

[2] GFEA (German Federal Environment Agency). 2007. Draft Dossier prepared in support of a proposal of endosulfan to be considered as a candidate for inclusion in the UN-ECE LRTAP protocol on persistent organic pollutants. German Federal Environment Agency. Umweltbundesamt, Berlin. http://www.unece.org/env/popsxg/docs/2007/Dossier_Endosulfan.2007.pdf

[3] *Hung H., Halsall C.J., Blanchard P., Li H., Fellin P., Stern G., Rosenberg B. Temporal trends of organochlorine pesticides in the Canadian Arctic atmosphere. Environ Sci Technol.;36(5):862-868 (2002)*

[4] *Bidleman, D.F., Cotham, W.E., Addison, R.F., Zinck, M.E. Organic contaminants in the Northwest Atlantic atmosphere at Sable Island, Nova Scotia 1988-89. Chemosphere 24, 1389-1412, (1992)*

Similar data on α-endosulfan have been reported from Resolute Bay (Cornwallis Island, 75 N lat.) where air concentrations of approximately 4 pg/m3 have been measured [43][1] and from air samples taken on an iceberg that calved off the Ward Hunt Ice Shelf on the northern shore of Ellesmere Island (approx. 81°N, 100°W). Mean concentration of α-endosulfan in summer 1986 and 1987 were 7.1 and 3.4 ng/m3, respectively [44][2]. Additional evidence for airborne long-range transport is provided by data from Newfoundland showing mean concentrations of 20 pg/m3 in summer 1977 [45][3].

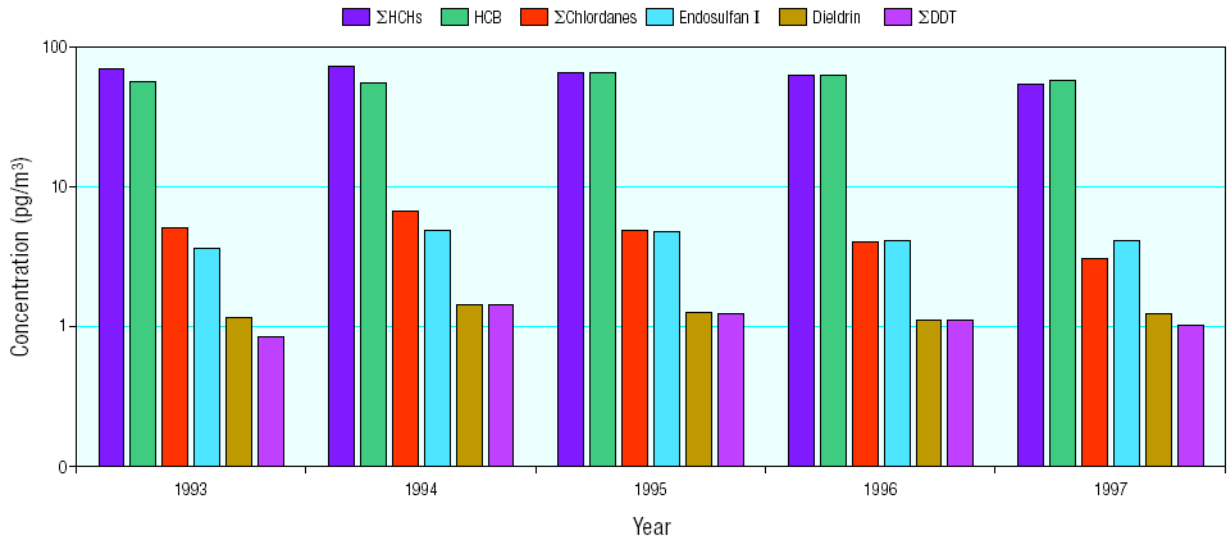

figure 4-2 Relative concentrations of selected organochlorine compounds at Alert, Nunavut, Canada (82°N, 42°W , source: [46]

Further air concentrations of endosulfan were reported from Amerma (eastern Arctic part of Russia) between 1–10 pg/m3 [47][4], [48][5]. Endosulfan was detected in around 90% of all samples displaying a significant correlation with atmospheric temperature. Unlike for other organochlorines with seasonal enhancements being suggested to be due to (re)volatilization from secondary sources, fresh applications were assumed to be responsible for endosulfan concentrations of 3.6 pg/m3 in winter and 5.8 pg/m3 in summer (mean values). Spatially, the annual concentrations at the various circumpolar sites did

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[1] Bidleman, T.F., R.L. Falconer, M.D. Walla. Toxaphene and other organochlorine compounds in air and water at Resolute Bay, N.W.T. Canada..Sci. Tot. Environ. 160/161, 55-63, (1995)

[2] *Patton, G.W., D.A. Hinckley, M.D. Walla, T.F. Bidleman. Airborne organochlorines in the Canadian High Arctic. Tellus, 41B, 243-255 (1989).*

[3] *Bidleman, T.F., E.J. Christensen, W.N. Billings. Atmospheric transport of organochlorines in the North Atlantic gyre. J. of Marine Research (39), 443-464, (1981)*

[4] *De Wit, C.A., A.T. Fisk, K.E. Hobbs, D.C.G. Muir. Levels, trends and effects of Persistent Organic Pollutants (POPs) in the Arctic environment. 2nd AMAP International Symposium on Environmental Pollution in the Arctic, Rovaniemi 1-3 October 2002*

[5] *Konoplev, A., P. Fellin, H. Li, P. Blanchrd, H. Hung, D. Samsonov, G. Stern Monitoring of POPs in Arctic Ambient Air: Initial results from Anderma (Russia) and Preliminary Assessment. 2nd AMAP International Symposium on Environmental Pollution in the Arctic, Rovaniemi 1-3 October 2002*

not show remarkable differences, indicating a degree of uniformity in contamination of the Arctic atmosphere.

Freshwater

Endosulfan (isomer unspecified) was measured also at Amituk Lake (75° 02′ 57′′N, 93° 45′51′′W) on Cornwallis Island, NV, Canada. The ranges were (in ng/L) 0,135 – 0.466 in 1992, 0.095 – 0.734 in 1993, and 0.217 – 0.605 in 1994 (quoted in [168][1]). Annual summertime peaks in endosulfan concentrations observed were attributed to fresh input from snow smelt via influent streams.

Seawater
Endosulfan was measured repeatedly in Arctic seawater during the 1990s. Mean concentrations were similar to those of chlordane and ranged from 2-10 pg/L [50]. Seasonal trends displayed increasing concentrations during the open water season suggesting fresh input from gas exchange and runoff. This trend parallels seasonal trends observed in Arctic air and Amituk Lake.

A survey of several pesticides in air, ice, fog, sea water and surface micro-layer in the Bering and Chuckchi Seas in summer of 1993 [51][2] identified α-endosulfan in air and subsurface seawater at levels around 2 pg/L. In melted ice less than 9 pg/L and for the sea water surface micro-layer less than 40 pg/L were detected. For fog condensates from several sites of that region concentration of <10 to <0.5 ng/L were reported. β-endosulfan was found in several atmospheric samples, e.g. from the Central Bering or Gulf of Anadyr at concentrations around 1 pg/m³. Similar concentrations of endosulfan have been reported from seawater samples from surface layer (40-60 m) collected in the Bering and Vhukchi Sea, north of Spitzbergen and the Greenland Sea [52][3].

Arctic seawater concentrations of two currently used pesticides, endosulfan and lindane were collected from 1990s to 2000 for different regions of the Arctic Ocean (Weber et al., 2006). Surface seawater concentrations for α- and β-endosulfan ranged from <0.1 to 8.8 pg L^{-1} and 0.1 to 7.8 pg L^{-1} respectively. Geographical distribution for α-endosulfan revealed that the highest concentrations in the western Arctic, specifically in Bering and Chukchi Seas with lowest levels towards the central Arctic Ocean. The results of air-water fugacity ratio indicate that α-endosulfan has been undergoing net deposition to surface waters across all the regions of the Arctic Ocean since 1990s. The authors concluded that the net deposition through air-water transfer may be the dominant pathway into the Arctic Ocean for α-endosulfan, particularly during the ice free periods.

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[1] *Ngabè, B., T.F. Bidleman. Endosulfan in the Atmosphere, Review and Evaluation. Report for Center of Coastal Environmental Health and Biomolecular Research, National Ocean Service, national Oceanic and Atmospheric Administration, Charleston, SC 29412, U.S.A. (2001)*

[2] Chernyak S.M., C.P. Rice, L.L. McConnell. Evidence of currently-used pesticides in air, ice, fog, seawater and surface. microlayer in the Bering and Chukchi Seas. Marine Pollution Bulletin 22 (5), 410-419, (1996)

[3] *Jantunen, T.F. Bidleman. Organochlorine Pesticides and Enantiomers of Chiral Pesticides in the Arctic Ocean Water. Arch. Environ. Contam. Toxicol. 35 218-228 (1998)*

Sediment
Laminated cores collected from Arctic Lake DV09 on Devon Island in May 1999 were analysed inter alia for endosulfan. Only α-endosulfan was present in the sediment of that lake. The concentration was highest at the sediment surface, and rapidly decreased to below detection limits in core slices dated prior to 1988 [53][1].

Snow and Snowpack
Endosulfan concentrations of α-endosulfan in snow samples collected in the Agassiz Ice Cap, Ellesmere Island, Canada in 1986 and 1987 were determined with a concentration range of 0.10 – 1.34 ng/m3 [54][2]. The concentrations of α-endosulfan in snowpack in Agassiz Ice Cap were 0.288 ng/L in 1989 and 0.046 ng/L in 1992 [55][3]. From measured snowpack concentrations and snowfall amounts winter deposition rates of 0.03 µg/m2 at minimum were estimated for the years 1986 and 1987 [56][4].

4.1.3.3 Biota
Blubber samples from male beluga (Delphinapterus leucas), collected over 20 years at five time points in Cumberland Sound, Canada. Only endosulfan sulfate was detected. But unlike other organochlorines levels appear to have increased steadily (3.2 fold) over that 20 year time period from 1982 to 2002 (ref. figure 4-3). α-endosulfan concentrations in blubber of minke whale (Balaenoptera acutorostrata) populations from distinct parts of the North Atlantic were sampled in 1998 [57][5]. The highest mean concentrations were found for whales in the North Sea/Shetland Islands (34 ng/g lipid for females and 43.0 ng/g for males), the Barents Sea (7.74 ng/g lipid for females and 9.99 ng/g for males) and Vestfjorden/ Lofotes (4.51 ng/g lipid for females and 9.17 ng/g lipid for males). Lower concentrations of < 1 ng/g and 5 ng/g lipid were reported for whales from Jan Mayen and Greenland. The differences were attributed to distinctions based on genetics, fatty acid profiles, etc.

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[1] *Stern, G.A., E. Braekevelt, P.A. Helm, T.F. Bideleman, P.M. Outridge, W.L.Lockhart, R. McNeeley, B.Rosenberg, M.G. Ikonomou, G.T. Tomy, P. Wilkinson Modern and historical fluxes of halogenated organic contaminants to a lake in the Canadian Arctic, as determined from annually laminated sediment cores. Env. Sci. Technol.*
[2] *Gregor, D.J., W. Gummer. Evidence of atmospheric transport and deposition of organochlorine pesticides and PCB in Canadian Arctic snow. Environ. Sci. Technol. 23 (5), 561-565 (1989)*
[3] *Franz, T.P., D.J. Gregor, S.J. Eisenreich. Snow deposition of atmospheric organic chemicals in: Baker, J.E. editor. Atmospheric deposition of contaminants to the Great Lakes and coastal waters. Pensacola, FL: Society for Environmental Toxicology and Chemistry 73-107 (1997)*
[4] *Barrie, L.A., D. Gregor, B. Hargrave, R. Lake, D. Muir, R. Shearer, B. Tracey, T. Bidleman Arctic contaminants: sources, occurrence and pathways. Sci. Tot. Environ. 122, 1-74 (1992)*
[5] *Hobbs, K.E., D.C.G. Muir, E.W. Born, R. Dietz, T. Haug, T. Metcalfe, C. Metcalfe, N. Øien Levels and patterns of persistent organochlorines in minke whale (Balaenoptera acutorostrata) stocks from the North Atlantic and European Arctic Environmental Pollution 121 (2), 239-252, (2003).*

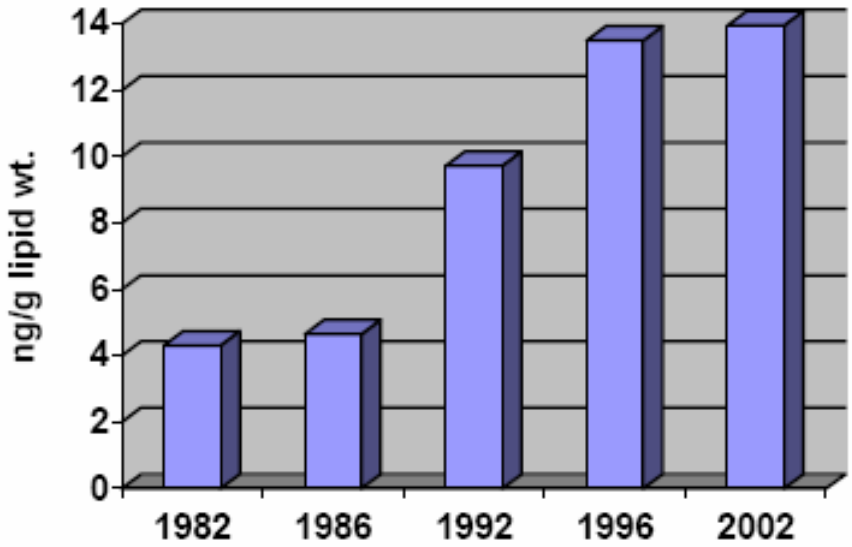

Figure 4-3 Temporal trends of age-adjusted concentrations of endosulfan in blubber of male beluga from Pangnirtung, Nunavut, Canada [58]

Endosulfan was also detected in adipose tissue and blood of polar bears from Svalbard. Mean values found for α-endosulfan were 3.8 ± 2.2 ng/g wet weight (min-max: 1.3-7.8 ng/kg) and 2.9 ± 0.8 ng/g for β-endosulfan (min-max: 2.2-4.3 ng/g) [59]. While the α-isomer was detectable in all samples (15/15) the β-isomer was found in just 5 out of 15 samples. In liver of northern fulmar (Fulmarus glacialis) from Bjørnøja endosulfans were detected for just two individuals out of fifteen at low levels of 0.28 and 0.50 ng/kg lipid weight [60][1].

Sediment

Laminated cores collected from Arctic Lake DV09 on Devon Island in May 1999 were analysed inter alia for endosulfan. Only α-endosulfan was present in the sediment of that lake. The concentration was highest at the sediment surface, and rapidly decreased to below detection limits in core slices dated prior to 1988 [53].

Biota
Endosulfan was also detected in adipose tissue and blood of polar bears from Svalbard. Mean values found for α-endosulfan were 3.8 ± 2.2 ng/g wet weight(min-max: 1.3-7.8 ng/kg) and 2.9 ± 0.8 ng/g for β-endosulfan (min-max: 2.2-4.3 ng/g)[59]. While the α-isomer was detectable in all samples (15/15) the β-isomer was found in just 5 out of 15 samples. In liver of northern fulmar (Fulmarus glacialis) from Bjørnøja endosulfans were

[1] *Gabrielsen G.W., L.B. Knudsen, M. Schlabach Organic Pollutants in Northern Fulmars (Fulmarius glacialis) from Bjørnøya SPFO-Report 922/2005, January 2005*

detected for just two individuals out of fifteen at low levels of 0.28 and 0.50 ng/kg lipid weight [60].

4.1.4 Modelling Data

Recent modelling data of EMEP Meteorolocical Synthesizing Centre East show that once released in Central Europe endosulfan (with endosulfan sulfat being not included), may spread out over the Northern Atlantic [65][1] reaching areas of Greenland. Thus its travelling distance is comparable to that of some other substances included in the POP Protocol, such as Benzo[a]pyrene.

[1] *N. Vulykh, E. Mantseva, V. Shatalov.Model assessment of potential for long-range transboundary atmospheric transport and persistence of Endosulfan EMEP Meteorological Synthesizing Centre East , Note 10/2005 (2005)*

6. ECOLOGICAL RISK CHARACTERIZATION

6.1 Risk Estimation

6.1.1 Post 2002 ERA Incident Data

EPA maintains a field incident database system (Ecological Incident Information System or EIIS) to track and evaluate nontarget plant and animal kills associated with pesticide use. The likelihood (certainty index) that a particular pesticide caused the incident is classified as "highly probable", "probable", "possible", or "unlikely", based on the information contained in the incident report. Since the 2002 RED chapter was issued where 91 incidents were reported, a total of 18 additional incidents have been reported (15 involving aquatic organisms and 3 involving terrestrial organisms) associated with the use of endosulfan. Specific details of the incidents are described in Table 6-1 and 6-2; 89% of the incidents were assigned a certainty index of "highly probable" to "probable" for endosulfan. Six of the incidents were the result of registered use, three were the result of misuse (intentional or accidental); it is unknown if the nine remaining incidents resulted from misuse or a registered use.

The aquatic incidences are more frequent compared to incidences involving terrestrial organisms. Typically, the number of individual aquatic organisms killed or adversely affected range from the hundreds to many thousands. California is the state with the most frequent reporting of newly found ecological incidences.

Table 6-1. EIIS Pesticide Summary Report: General Information- Endosulfan (079401)

Date	Incident #	Use Site	County	State	Certainty	Legal.	Formul.	Appl. Method	Total Magnitude
AQUATIC									
6/10/1996	I003668-001	Agricultural Area	Rapides	LA	4	RU		N/R	6500
6/19/1996	I004668-003	N/R	Rapides	LA	4	UN			500
6/29/1996	I004993-010	N/R	PLACER	CA	3	UN		N/R	200
7/1/1996	I003659-001	Tomato	ACCOMACK	VA	3	UN		Spray	THOUSANDS
7/15/1996	I004993-011	N/R	IMPERIAL	CA	4	UN		N/R	3000
7/15/1996	I004864-001	ALFALFA	Imperial	CA	4	MI		AERIAL	5000
8/4/1996	I004439-069	ALFALFA		CA	3	RU			THOUSANDS
6/16/1997	I007546-050	Agricultural Area		IN	3	MA	N/R	Spray	UNKNOWN
10/2/1997	I006173-001	Agricultural Area		TX	3	RU	N/R	N/R	UNKNOWN
7/23/1998	I012265-002	Potato		PE	3	RU	N/R	N/R	UNKNOWN
8/9/2000	I017028-001	Potato		PE	2	UN			>50
9/1/2000	I012283-001	Agricultural Area	Sequatchie	TN	3	MA		Spill	200000
9/10/2002	I014189-001	Cotton	Riverside	CA	4	RU		Spray	650
10/14/2003	I014884-022	N/R	Kings	CA	3	UN			various species
6/21/2006	I018075-001	N/R	Imperial	CA	4	UN			5000
TERRESTRIAL									
1/1/1999	I010533-001	Cotton	3	RU	N/R	N/R	UNKNOWN		
7/7/2001	I012973-001	N/R	Monroe	NY	1	UN			1
1/14/2002	I012626-001	N/R	Montgomery	MD	3	UN			various animals

Certainty Code: 0=Unrelated, 1=Unlikely, 2=Possible, 3=Probable, 4=Highly Probable.

Legality Code: RU=Registered Use, M=Misuse, MA=Misuse (Accidental), MI=Misuse (Intentional), U=Unknown.

Table 6-2. EIIS Pesticide Summary Report: Species Information- Endosulfan (079401)

Date	Incident #	Species	Scientific Name	Magnitude	Response	Rt. Exposure
AQUATIC						
6/10/1996	I003668-001					
		bass	Centrarchidae spp.	hundreds	mortality	Runoff
		bowfin	Amia calva	hundreds	mortality	Runoff
		carp	Cyprinus carpio	hundreds	mortality	Runoff
		channel catfish	Ictalurus punctatus	hundreds	mortality	Runoff
		crappie	Centrarchidae	hundreds	mortality	Runoff
		flathead catfish	Pylodictis oilvaris	hundreds	mortality	Runoff
		shad	Clupeidae	hundreds	mortality	Runoff
6/19/1996	I004668-003					
		bowfin	Amia calva	some of 500	mortality	Flowing water
		carp	Cyprinus carpio	some of 500	mortality	Flowing water
		crappie	Centrarchidae	some of 500	mortality	Flowing water
		shad	Clupeidae	some of 500	mortality	Flowing water
6/29/1996	I004993-010					
		trout	Salmonidae	200	mortality	N/R
7/1/1996	I003659-001					
		clam	Bivalvia	thousands	mortality	Runoff
7/15/1996	I004864-001					
		carp	Cyprinus carpio	thousands	mortality	
		shad	Clupeidae	thousands	mortality	
		tilapia	Oreochreomis aureu	thousands	mortality	
7/15/1996	I004993-011					
		carp	Cyprinus carpio	thousands	mortality	N/R
		shad	Clupeidae	thousands	mortality	N/R
		tilapia	Oreochreomis aureu	thousands	mortality	N/R
8/4/1996	I004439-069					
		unknown fish		thousands	mortality	N/R
6/16/1997	I007546-050					
		n/r		unknown	mortality	Ingestion
		turtle	Testudines	1	mortality	Ingestion

10/2/1997	I006173-001					
		n/r		unknown	mortality	Runoff
7/23/1998	I012265-002					
		n/r		unknown	mortality	Runoff
8/9/2000	I017028-001					
		stickleback	Gasterosteiformes	unknown	mortality	Runoff
		trout	Salmonidae	>50	mortality	Runoff
9/1/2000	I012283-001					
		darter	Etheostoma sp.	thousands	mortality	Flowing water
		largemouth bass	Micropterus salmoides	thousands	mortality	Flowing water
9/10/2002	I014189-001					
		channel catfish	Ictalurus punctatus	at least 1	mortality	Runoff
		striped bass	Morone saxatilis	600-700	mortality	Runoff
		threadfin shad	Dorosoma petenense	at least 4	mortality	Runoff
		tilapia	Oreochreomis aureu	at least 2	mortality	Runoff
10/14/200	I014884-022					
		bullhead	Ameiurus sp.	over 50	mortality	Flowing water
		carp	Cyprinus carpio	over 100	mortality	Flowing water
		threadfin shad	Dorosoma petenense	over 200	mortality	Flowing water
6/21/2006	I018075-001					
		carp	Cyprinus carpio	most of 5,000	mortality	Flowing water
		catfish	Ictaluridae		mortality	Flowing water
		redhorse	Moxostoma sp.		mortality	Flowing water
TERRESTRIAL						
1/1/1999	I010533-001					
		frog	Anura	1	mortality	Ingestion
		owl	Strigidae	1	mortality	Ingestion
		termite	Isoptera		mortality	Ingestion

7/7/2001	I012973-001					
		cooper's hawk	Accipiter cooperii	1	mortality	Ingestion
1/14/2002	I012626-001					
		blue jay	Cyanocitta cristata	1	mortality	N/R
		crow	Corvus sp.	1	mortality	N/R
		opossum	Didelphimorphia	1	mortality	N/R
		red fox	Vulpes fulva	1	mortality	N/R
		squirrel	Sciuridae	12	mortality	Ingestion

6.1.2 Aquatic Organisms: Water Column Exposure

Results from the aquatic exposure modeling of total endosulfan (α+β+sulfate) in surface water combined with the toxicity information for endosulfan indicate that freshwater and saltwater fish and invertebrates are potentially at risk from endosulfan application at the allowable maximum label rates. Risk quotients (ratios of estimated exposure concentrations to the most sensitive toxicity endpoint) for freshwater and saltwater organisms are presented in Tables 6-3 and 6-4, respectively, for the Florida tomato and California strawberry crop scenarios. The Florida tomato crop scenario was chosen because it resulted in the highest EECs relative to other crop scenarios modeled in the 2002 ERA. These risk quotients reflect new information on both the toxicity of endosulfan (i.e., comparative toxicity of degradate, endosulfan sulfate) and the surface water exposure assessment (modeling of total endosulfan concentrations in water, including α, β and endosulfan sulfate). The addition of new toxicity information for endosulfan sulfate for water column exposures did not change the most sensitive toxicity values used to calculate risk quotients. However, PRZM/EXAMS modeling of the total endosulfan concentrations did result in modest increases in EEC values compared to the 2002 ERA. Specifically, the peak, 21-d and 56-d average EECs for endosulfan (α+β) increased by 21%, 41% and 39%, respectively, compared to the EECs for α + β endosulfan from the 2002 ecological risk assessment. Because new information did not change the toxicity values used to derive RQ values from the 2002 ERA, RQ values increased by the same magnitude as the EECs. That is, acute RQ values (fish & invertebrate), chronic RQ (invertebrate), and chronic RQ (fish) increased by 21%, 41% and 39% with the addition of endosulfan sulfate to the EEC values.

Consistent with the 2002 ERA, the RQ values exceeded level of concern (LOC) for acute toxicity to fish (fresh and saltwater) and invertebrates (fresh and saltwater) with the Florida crop scenario. The RQ values also exceeded fish and invertebrate LOCs with the California strawberry crop scenario. Maximum acute and chronic freshwater RQ values are **28** for fish and **133** for invertebrates. Maximum acute and chronic saltwater RQ values are **230** for fish and **680** for invertebrates.

Table 6-3. Acute and chronic risk quotients for freshwater fish (rainbow trout *Oncorhynchus mykiss*) and invertebrates (scud *Gammarus lacustris*) exposed to endosulfan (2002 ERA vs. 2007 Addendum)

Annual Maximum Crop Application Rate (# of apps)	EECs	Acute Risk Quotients		Chronic Risk Quotients	
	Peak / 21-day Average 56-day Average (ug/L)	Freshwater Fish LC_{50} = 0.83 µg/L	Freshwater Invertebrate LC_{50} = 5.8 µg/L	Freshwater Fish NOEC = 0.11 µg/L	Freshwater Invertebrate NOEC = 0.07 µg/L
2002 ERA:					
Florida Tomato	19	23 [(a)]	3.3 [(a)]	–	–
3.0 (3)	6.5	–	–	--	93 [(b)]
	4.9	--	--	44 [(b)]	--
2007 Addendum:	23	**28** [(a)]	**4.0** [(a)]	--	--
Florida Tomato	9.3	–	--	--	**133** [(b)]
3.0 (3)	6.8	--	--	**62** [(b)]	--

Table 6-3. Acute and chronic risk quotients for freshwater fish (rainbow trout *Oncorhynchus mykiss*) and invertebrates (scud *Gammarus lacustris*) exposed to endosulfan (2002 ERA vs. 2007 Addendum)

Annual Maximum Crop Application Rate (# of apps)	EECs	Acute Risk Quotients		Chronic Risk Quotients	
	Peak / 21-day Average 56-day Average (ug/L)	Freshwater Fish LC_{50} = 0.83 µg/L	Freshwater Invertebrate LC_{50} = 5.8 µg/L	Freshwater Fish NOEC = 0.11 µg/L	Freshwater Invertebrate NOEC = 0.07 µg/L
2007 Addendum:					
California	12	**15** (a)	**2.1** (a)	--	--
Strawberry	5.5	–	--	--	**79** (b)
3.0 (3)	3.9	--	--	**36** (b)	--

Table 6-4. Acute and chronic risk quotients for estuarine/marine fish (stripped bass *Morone saxatilis*) and invertebrates (Eastern oyster *Crassostrea virginica*) exposed to endosulfan (2002 ERA vs 2007 Addendum)

Annual Maximum Crop Application Rate (# of apps)	EECs	Acute Risk Quotients		Chronic Risk Quotients	
	Peak 21-day Average 56-day Average µg/L	Estuarine/ marine Fish LC_{50} = 0.1 µg/L	Estuarine/ marine Invertebrate LC_{50} = 0.45 µg/L	Estuarine/ marine Fish NOEC = 0.01 µg/L	Estuarine/ marine Invertebrate NOEC = 0.05 µg/L
2002 ERA:	19	191(a)	42 (a)	–	–
Tomato	6.5	–	–	--	130 (b)
3.0 (3)	4.9	--	--	487 (b)	--
2007 Addendum	23	**230** (a)	**51**(a)	--	--
Tomato	9.3	--	--	--	**186** (b)
3.0 (3)	6.8	--	--	**680** (b)	--
2007 Addendum:					
California	12	**120** (a)	**27** (a)	--	--
Strawberry	5.5	–	--	--	**110** (b)
3.0 (3)	3.9	--	--	**390** (b)	--

6.1.3 Aquatic Organisms: Sediment Exposure

Results from the aquatic exposure modeling of total endosulfan (α+β+sulfate) in sediment pore water combined with the toxicity information for endosulfan indicate that freshwater and estuarine invertebrates are potentially at risk from endosulfan application at the allowable maximum label rates. Risk quotients (ratios of estimated exposure concentrations to the most sensitive toxicity endpoint) derived for predicted sediment pore water concentrations are provided in Table 6-5 for the Florida tomato and California strawberry crop scenarios. The calculated RQ values ranged from **1.6** to **2.8** for the California strawberry and Florida tomato crop scenarios, respectively for estuarine.

Table 6-5. Risk Quotients for Aquatic Invertebrates to Endosulfan via Sediment Pore Water Exposure

Exposure Scenario	21-d Average Pore Water EEC (µg/L)	Freshwater Pore Water RQ [(1)]	Estuarine/Marine Pore Water RQ [(2)]
Florida Tomatoes	4.4	**1.6**	**2.8**
California Strawberries	2.5	0.9	**1.6**

[(1)] RQs for freshwater invertebrates based on 10-d NOAEC of 2.7µg/L for endosulfan (as endosulfan sulfate) in sediment pore water for the midge, *Chironomus tentans* (MRID 463826-05).
[(2)] Risk quotients for estuarine/marine invertebrates calculated using a 28-d chronic NOAEC of 1.58 µg/L for endosulfan (as endosulfan sulfate) in sediment pore water for the estuarine amphipod, *Leptocheirus plumulosus* (MRID 469290-01).
Values in bold indicate exceedence of chronic LOC of 1.

6.14 Terrestrial Organisms: Piscivorous Wildlife

In order to assess risks to mammals and birds consuming aquatic organisms associated with predicted endosulfan concentrations in aquatic organisms, several species were selected, including mink *(Mustela vison)*, river otter *(Lutra canadensis)*, belted kingfisher *(Ceryle alcyon)*, herring gull *(Larus argentatus)*, osprey *(Pandion haliaetus)*, mallard duck *(Anas platyrhynchos)*, great blue heron *(Ardea herodias)* and bald eagle *(Haliaeetus leucocephalus)*. Information on species ecological and physiological characteristics (dietary composition, body weights, food consumption rates, drinking water rates, etc) is provided in Attachment B. Endosulfan toxicity data used in this comparison are shown in Table 6-6.

Table 6-6. Summary of toxicity of endosulfan to mammals and birds. Bold indicates parameters used for RQ derivation.

Species	Endpoint	Value (ppm)	MRID
Laboratory rat (*Rattus norvegicus)*	LD_{50}	**10**	0038307
	NOEC*	**15**	00148264
Northern bobwhite quail *(Colinus virginianus)*	LC_{50}	**805**	22923
	NOEC	60	40261303
Mallard duck *(Anas platyrhynchos)*	LD_{50}	**28**	136998
	LC_{50}	1053	22923
	NOEC**	**30**	40261302

*Effected endpoint at LOEC: growth
**Effected endpoint at LOEC: growth and reproduction

Following standard OPP avian and mammalian risk assessment practices, risk quotients (RQ) were calculated using exposures expressed on an ingested dose basis (mg/kg-bw/d) and a dietary basis (mg/kg-diet). Exposure concentrations in the diet of piscivorous wildlife were predicted using an aquatic food web bioaccumulation model as described in Section 2.2 and Attachment B. The RQ values are then compared to Agency Levels of Concern (LOC) for determination of potential risk. For acute exposures, RQ values associated with mean predicted concentrations in aquatic biota exceed the Agency acute risk LOC (0.1) for one of the eight species modeled (river otter, Table 6-7). At higher percentiles of predicted exposure concentrations, exceedences of the acute LOC also occur mink and belted kingfisher. Although the acute LOC of 0.1 is exceeded for these species, all RQ values are less than 0.4 or less, indicating the magnitude of risk is relatively small and likely to be sensitive to modeling assumptions. Calculated RQ

values resulting from chronic exposure to predicted endosulfan concentrations in aquatic biota are shown in Table 6-8. Results indicate RQ values are all below the Agency's LOC of 1.0 for chronic risks.

Table 6-7. Predicted RQ Values for Piscivorous Mammals and Birds Exposed to Endosulfan Through Acute, Dose-based Exposures. (All parameters varied according to Table 10 in Attachment B).
All dietary-based RQ values for birds are <0.01.

Organism	Mean	SD	25th %	75th %	90th %
Dose-Based					
Mink	0.07	0.08	0.02	0.09	**0.18**[1]
River otter	**0.15**[1]	**0.25**[1]	0.04	**0.20**[1]	**0.39**[1]
Belted kingfisher	0.08	0.08	0.02	**0.11**[1]	**0.20**[1]
Herring gull	0.03	0.05	0.01	0.04	0.07
Osprey	0.03	0.03	0.01	0.03	0.07
Mallard duck	0.02	0.02	0.01	0.03	0.05
Great blue heron	0.02	0.02	0.01	0.03	0.05
Bald eagle	0.01	0.02	<0.01	0.02	0.03

[1] **Exceeds LOC (0.1) for acute exposures to listed animals.**

Table 6-8. Predicted RQ values for mammals and birds exposed to endosulfan through chronic, dose- and dietary-based exposures. All parameters varied according to Table 10 in Attachment B.

Organism	Mean	SD	25th %	75th %	90th %
Dose-Based					
Mink	0.03	0.03	0.01	0.04	0.08
River otter	0.06	0.11	0.02	0.08	0.16
Dietary-based					
Mink	0.04	0.04	0.01	0.05	0.09
River otter	0.37	0.61	0.10	0.49	0.95
Belted kingfisher	0.01	0.01	<0.01	0.01	0.02
Herring gull	0.03	0.04	0.01	0.04	0.07
Osprey	0.04	0.04	0.01	0.05	0.09
Mallard duck	0.02	0.02	0.01	0.02	0.04
Great blue heron	0.05	0.05	0.01	0.07	0.12
Bald eagle	0.07	0.11	0.02	0.09	0.17

6.1.5 Nontarget Terrestrial Organisms: Herbivorous and Insectivorous Wildlife

The addition of new information on the toxicity of endosulfan sulfate did not alter the risk quotient calculations for nontarget terrestrial organisms (herbivorous and insectivorous birds and mammals). For completeness, however, the RQ values from the 2002 ERA (USEPA, 2002) are included here for the crop application scenario that produced the highest RQ values.

The estimated environmental concentration (EEC) values used for terrestrial exposure are derived from the Kenaga nomograph, as modified by Fletcher *et al.* (1994), based on a large set of actual field residue data. The upper limit values from the nomograph represent the 95th percentile of residue values from actual field measurements (Hoerger and Kenega, 1972). The Fletcher et al. (1994) modifications to the Kenaga nomograph are based on measured field residues from 249 published research papers, including information on 118 species of plants, 121 pesticides, and 17 chemical classes. These modifications represent the 95th percentile of the expanded data set. Risk quotients are based on the most sensitive LC_{50} and NOAEC for birds (in this instance, mallard ducks and bobwhite quail) and LD_{50} for mammals (based on lab rat studies).

Acute and chronic risk quotients were calculated following the procedure outlined in Appendix F of the 2002 ERA (USEPA, 2002) and were then compared to LOCs. Acute high risk, restricted use and endangered species LOCs are exceeded for birds (Table 6-9) and mammals (Table 6-10) at label application rates for the major crops modeled (maximum acute RQs of 0.53 and 40 for birds and 15-g mammal consuming short grass). Chronic LOCs for birds (Table 6-9) were also exceeded, with a maximum RQ of 2.7. Chronic LOCs for mammals (Table 6-11) were exceeded by a maximum RQ of 5.4 on short grass.

Table 6-9. Avian acute and chronic risk quotients for a single and multiple broadcast applications of nongranular products of endosulfan based on a bobwhite quail LC_{50} of 805 ppm and a mallard duck NOEC of 30 ppm.

Use/App. Method	Rate (lbs ai/A) x No. Apps. (Interval, da)	Food Items	Max. EEC (mg/kg)[e]	Avg. EEC (mg/kg)[e]	Acute RQ ($EEC/_{LC50}$)	Chronic RQ (EEC/ NOAEC)
Apples (air blast), grapes (aerial), pecans (air blast)	1.5 lbs./A (2) 10-day interval	Short grass	424	81	0.53[a]	2.7[d]
		Tall grass	194	34	0.24[b]	1.1[d]
		Broadleaf plants/Insects	238	39	0.30[b]	1.3[d]
		Seeds	26	4	0.03	0.13

Source: Table 11 of 2002 ERA (USEPA, 2002)

[a] exceeds acute high, acute restricted and acute endangered species LOCs.

[b] exceeds acute restricted and acute endangered species LOCs.

[c] exceeds acute endangered species LOCs

[d] exceeds chronic LOC

[e] estimated environmental concentrations predicted using 1st-order degradation model based on foliar dissipation.

Table 6-10. Acute RQ values for small (15 g), intermediate (35 g) and large (1,000 g) mammals feeding on short or tall grass, broadleaf plants/insects, and seeds exposed to endosulfan following single and multiple applications.

Site (method) Application Rate (number of applications)	Body Weight, g	RQ Short Grass	RQ Tall Grass	RQ Broadleaf Plants/Insects	RQ Seeds

Table 6-10. Acute RQ values for small (15 g), intermediate (35 g) and large (1,000 g) mammals feeding on short or tall grass, broadleaf plants/insects, and seeds exposed to endosulfan following single and multiple applications.

Site (method) Application Rate (number of applications)	Body Weight, g	RQ Short Grass	RQ Tall Grass	RQ Broadleaf Plants/Insects	RQ Seeds
apples (air blast), grapes (aerial), pecans (air blast) 1.5 lbs. a.i./A (2)	15	40 [a]	18 [a]	23 [a]	0.55 [a]
	35	28 [a]	13 [a]	16 [a]	0.39 [b]
	1000	6.3 [a]	2.9 [a]	3.6 [a]	0.08

Source: Table 12 of 2002 ERA (USEPA, 2002)
[a] exceeds acute high, acute restricted and acute endangered species LOCs.
[b] exceeds acute restricted and acute endangered species LOCs.

Table 6-11. Chronic RQ values for mammals feeding on short grass, tall grass, broadleaf plants/insects, and seeds exposed to endosulfan following multiple applications.

Site (method) Application Rate (number of applications)	RQ Short Grass	RQ Tall Grass	RQ Broadleaf Plants/Insects	RQ Seeds
apples (air blast), grapes (aerial), pecans (air blast) 1.5 lbs. a.i./A (2)	5.4 [a]	2.3 [a]	2.6 [a]	0.3

Source: Table 13 of 2002 ERA (USEPA, 2002)
[a] exceeds chronic LOC

6.2 Risk Conclusions

Risk quotient (RQ) predicted for aquatic organisms resulting from water column exposure to endosulfan are about 20% to 40% higher compared to the 2002 ERA. This increase reflects the addition of endosulfan sulfate to the exposure modeling and data indicating it is of similar toxicity to the parent isomers (α and β). Based on the tomato crop scenario that yielded the largest EECs, acute and chronic RQ values range from about 30 to 60 for freshwater fish, respectively (compared to 23 and 44 from the 2002 ERA) to about 230 and 680 for estuarine/marine fish, respectively (compared to 190 and 490 from the 2002 ERA). Acute and chronic RQ values for invertebrates range from approximately 4 to 130 for freshwater (compared to 3.3 and 93 from the 2002 ERA) and from 50 to 190 for estuarine/marine (compared to 42 to 130 from the 2002 ERA), respectively. Findings from ecological incidents support the findings of endosulfan risk to fish and invertebrates.

Risks to freshwater and estuarine/marine invertebrates resulting from sediment exposure to endosulfan are evident from the integration of exposure and effect characterization. The RQ values for sediment-dwelling invertebrates range form 0.9 to

2.8 depending on species. These RQ values are based on predicted total endosulfan residues compared to endosulfan sulfate toxicity values, which assumes similar toxicity of endosulfan sulfate and total endosulfan residues.

Based on preliminary results from an aquatic food web bioaccumulation model, risks to piscivorous wildlife appear relatively modest with mean predicted acute RQ values exceeding the Agency acute LOC of 0.1 for one of eight species modeled (0.15 for river otter) and 90^{th} percentile estimates exceeding the LOC for three of eight species modeled (0.18, 0.39, 0.20 for mink, river otter and belted kingfisher, respectively). Predicted chronic RQ values did not exceed the Agency LOC for any of the eight species modeled.

Risks to nontarget terrestrial wildlife did not change from the 2002 ERA as a result of this addendum, because currently available terrestrial exposure models could not address total residue exposure. Based on the 2002 ERA, RQs for birds and mammals exceed the Agency's acute and chronic risk LOCs and range up to a maximum of 2.7 for birds and 40 for mammals.

In summary, there is a concordance of evidence that endosulfan is undergoing long-range transport and has moved to sites distant from use areas. Additionally, while the parent may readily undergo degradation under some environmental conditions, the sulfate degradate is persistent and represents a source for endosulfan to enter aquatic and terrestrial food chains. While endosulfan is not expected to biomagnify appreciably in aquatic food webs, the compound does bioconcentrate in aquatic organisms to a significant extent. Also, there is direct evidence (measured residues) that endosulfan bioaccumulates in terrestrial systems and indirect evidence (modeling) that endosulfan has a significant potential to biomagnify in certain terrestrial food webs. Monitoring data and incident reports confirm that endosulfan is moving through aquatic and terrestrial food chains and that its use has resulted in adverse effects on the environment adjacent to and distant from it registered use sites.

References

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ATTACHMENT A.

REVIEW OF EMPIRICAL BIOACCUMULATION DATA FOR ENDOSULFAN

The purpose of this review is to provide a preliminary indication of how the Agency's understanding of the bioaccumulation potential of endosulfan might change as a result of new information being considered since the publication of EPA's 2002 Endosulfan Ecological Risk Assessment (EPA, 2002). This review is considered preliminary for two reasons. First, it is not intended to be comprehensive. This literature review was focused on controlled experiments of endosulfan bioconcentration or bioaccumulation rather than field studies of the distribution of endosulfan in various environmental compartments. The scope was constrained in this way primarily because of practical limitations (time constraints) and also the expectation that biomagnification of endosulfan (and degradates) would not likely be a major factor given its moderate hydrophobicity (log K_{ow} 3-4.5). Controlled laboratory studies of bioconcentration generally involve less uncertainty in quantifying chemical exposure by organisms and thus, generally contain less uncertainty in calculated BCFs compared to field studies. Second, the available data were not subjected to formal data evaluation procedures (e.g., Data Evaluation Records), again, due to time and resource constraints.

I. OVERVIEW OF EMPIRICAL BIOACCUMULATION STUDIES

Tables A-1 and A-2 summarize the available studies reviewed on the bioconcentration and bioaccumulation of endosulfan by aquatic organisms. Bioconcentration refers to the net accumulation of a chemical by an organism that results from exposure through water only, usually through uptake across gills, other respiratory surfaces or integument. Bioaccumulation is the net accumulation of a chemical by an organism that results from exposure to all environmentally relevant exposure routes, including water, food, sediment, etc. (U.S. EPA, 2003). For certain persistent, highly hydrophobic organic chemicals (usually log K_{ow} >5), non-aqueous exposure can become a significant exposure route to aquatic organisms and result in higher concentrations in organisms compared to uptake from water alone (bioconcentration).

Studies on the bioconcentration and bioaccumulation of endosulfan and its degradates were identified using the EPA ECOTOX database (http://cfpub.epa.gov/ecotox/). Other studies were identified through targeted literature searches and secondary reviews. Individual studies were obtained and reviewed informally for data quality and results summarized accordingly. In general, data quality was evaluated against three main criteria:

- Achievement of steady state conditions,
- Stability of exposure concentrations over time, and
- Quantification of relevant endosulfan constituents (α and β isomers and the primary metabolite endosulfan sulfate).

Of the 11 studies evaluated, none satisfied all three of the above criteria completely. Therefore, an important aspect of this preliminary review is presenting a clear characterization of the limitations and uncertainty in the available bioaccumulation data.

A. Bioconcentration/Bioaccumulation in Fish

Overall range in BCF Values. Bioconcentration studies were identified and reviewed for seven species of fish, including , sheepshead minnow (*Cyprinodon variegatus*), zebra fish (*Danio rerio*), yellow tetra (*Hyphessobrycon bifasciatus*), striped mullet (*Mugil cephalus*), pinfish (*Lagodon rhomboides*), long whiskers catfish (*Mystus gulio*) and spot (*Leiostomus xanthurus;* Table A-1). No obvious difference was evident between bioconcentration by estuarine/marine and freshwater fish, although data were very limited to support definitive comparisons. For fish, the reported BCF values ranged from approximately 20 to 11,600 (L/kg wet wt.[23]). With the exception one species (yellow tetra), BCFs were less than 3,000 for the remaining six fish species. However, as discussed further below, when the uncertainty in the underlying BCF data is considered, the actual range in BCFs for fish is unclear.

Depuration Rate/Half Life. Based the data from three studies where endosulfan depuration was evaluated, the depuration rate (biological half life) of endosulfan in fish appears to be relatively fast (approximately 2-6 days for zebra fish, yellow tetra, and striped mullet; Toledo and Jonsson, 1992; Jonsson and Toledo, 1993; Schimmel *et al*., 1977). Interestingly, steady-state conditions did not appear to be reached in yellow tetra and striped mullet after 21 and 28 days, respectively. This long time to reach steady state is not expected given the short biological half lives reported by these authors (approximately 2 days for total endosulfan measured as α+β+sulfate). The reason for this apparent inconsistency is unclear, but may indicate that endosulfan accumulation kinetics is more complicated than a simple first-order phenomenon.

BCF Study Quality/Data Interpretation. It is of critical importance to consider the limitations and uncertainties in the BCF values presented in Table A-1. Detailed reviews of each study are provided in Section II. Based on these reviews, none of the data meet all three primary evaluation criteria: (1) existence of steady-state conditions, (2) stability of exposure concentrations over time, and (3) quantification of endosulfan parent compounds and metabolites. A summary of how the fish BCF studies compared against these criteria is provided below.

- **Steady-State Conditions.** The existence of steady state between chemical concentration in organisms and their exposure media (water for BCF) is important because a steady-state BCF reflects the highest, long-term bioconcentration potential by an organism. Thus, a BCF estimated when an organism has not achieved steady state with its surrounding exposure concentrations may underestimate or in some cases, overestimate long-term bioconcentration potential. Based on the study reviews presented in Section III and summarized in

[23] Unless otherwise noted, all BCF and BAF values discussed herein are expressed as L/kg wet weight.

Table A-1, none of the studies that used the ratio method for determining a BCF for fish established that steady-state conditions were achieved. Most of the studies only evaluated endosulfan accumulation at test termination, and therefore, steady-state conditions could not be evaluated. The study by Schimmel *et al* (1977) is the only fish BCF study that used the ratio method and reported accumulation data over time (28-d striped mullet study). However, increases in tissue concentrations of total endosulfan (α, β, sulfate) towards the end of this study (day 21 and 28) suggest that steady-state conditions may not have been reached. If this is the case, than the BCF of 2,755 L/kg w.w. may underestimate long-term bioconcentration of total endosulfan by striped mullet.

By definition, BCFs determined by the kinetic method reflect steady-state conditions (assuming that first order accumulation kinetics apply and exposure concentrations remain relatively constant). Two steady-state kinetic BCFs were available for two species (zebra fish and yellow tetra) using identical study protocols (Toledo and Jonsson 1992; Jonsson and Toledo, 1993). The reported kinetic-based BCF for zebra fish (2,650) is similar to the 21-d BCF calculated by this reviewer using residue data reported by the authors (2,680), which supports the reported kinetic-based BCF value. However, the kinetic-based BCF value of 11,583 derived by Jonsson and Toledo (1993) for yellow tetra (based on α, β and endosulfan sulfate) is questionable. Specifically, the accumulation pattern for yellow tetra reported by Jonsson and Toledo indicated that steady state was not reached by 21 days, which is inconsistent with the elimination rate and calculated biological half life of approximately 2 days reported by the authors. This inconsistency raises questions regarding the accuracy of the kinetic-based BCF reported for yellow tetra. If a ratio method is applied based on the observed accumulation in fish and estimated nominal concentration in water, a 21-d non-steady state BCF of 5,670 is calculated. As noted below, the studies with zebra fish and yellow tetra used a static-renewal exposure system whereby exposure concentrations were assumed to drop by 50% between renewal periods which adds uncertainty to these BCF values.

Stability of Exposure Concentrations. Of the five fish BCF studies reviewed, stability of exposure concentrations was documented in two studies (Hansen and Cripe, 1991; Schimmel *et al*., 1977). Both these studies used a flow-through exposure system (typically required for bioconcentration studies according to ASTM and OPP guidelines) and measured concentrations in exposure water. The remaining three BCF studies with fish either did not measure exposure concentrations (Toledo and Jonsson, 1992; Jonsson and Toledo, 1993) or provided insufficient information to evaluate variability in exposure concentrations over time (Rajendran and Venugopalan, 1991).

Quantification of Parent Compound and Metabolites. The most accurate assessment of endosulfan bioaccumulation should include measurement of both isomers (α, β) and its principle metabolite, endosulfan sulfate in tissue. Of the five fish BCF studies evaluated, one (Schimmel *et al*., 1977) quantified α, β, and

endosulfan sulfate in water and organisms while two studies (Toledo and Jonsson, 1992; Jonsson and Toledo, 1993) quantified these compounds in organisms only. The study by Hansen and Cripe (1991) quantified the α and β isomers only while that by Rajendran and Venugopalan (1991) did not report which endosulfan constituents were quantified.

Bioaccumulation Studies with Fish. No data on the bioaccumulation of endosulfan in fish (i.e., uptake from multiple exposure routes) were found, but as discussed previously, a comprehensive review of field data on endosulfan bioaccumulation was not conducted.

B. Bioconcentration/Bioaccumulation by Invertebrates

Overall Range in BCF/BAF values. Bioconcentration studies with aquatic invertebrates were available for five species of invertebrates and included the blue mussel (*Mytilus edulis),* grass shrimp, (*Palaemonetes pugio),* oyster, (*Crassostrea madrasensis)*, clam, (*Katelysia opima)* and red swamp crayfish, (*Procambarus clarkii).* Based on the studies presented in Table A-1, the bioconcentration of endosulfan in aquatic invertebrates appears to be lower than those reported for fish, ranging from about 20 to 600 (L/kg w.w.). The value of 1.9 from Naqvi and Newton (1990) for crayfish is considered highly suspect and is not considered further (see Sections II and III). Bioaccumulation studies (i.e., those that included exposure to multiple uptake routes) were available for three invertebrates, including the mussel (*Mytilus galloprovincialis)*, Eastern oyster, (*Crassostrea virginica)*, and the water flea, (*Daphnia magna*; Table A-2). Bioaccumulation factors (Table A-2) for the Eastern oyster and *D. magna* for total endosulfan are approximately 600. In a short-term study by DeLorenzo *et al* (2002), uptake of endosulfan from food (contaminated algae) by *D. magna* was documented as negligible compared to uptake from the water column.

Depuration Rate/Half Life. Information on the depuration of endosulfan by invertebrates was only available for the blue mussel. In one study, Ernst (1977) reported a depuration half life of 33.8 hours (about 1.5 days). In a long-term study, Roberts (1972) reported that concentrations declined rapidly in the blue mussel after two weeks for two of the three exposure concentrations (500 and 1000 µg/L, but declined more slowly in the lowest exposure concentration (100 µg/L). Depuration data were not tabulated by Roberts (1972) but based on the graphical representation of the depuration data, the depuration half life appears to approximate two weeks. It is important to note that both the Ernst (1977) and Roberts (1972) study of endosulfan depuration in blue mussels have several significant limitations which add uncertainty in the half life determinations. Both studies do not report which endosulfan constituents were present in the tissue residue analysis. Furthermore, in the 122-d study (Roberts, 1972), mussels were losing tissue mass over time which can lead to misleading conclusions regarding chemical accumulation when evaluated on a concentration basis (*i.e*., concentrations in tissue can increase solely because tissue mass decreases).

Study Quality/Data Interpretation. A number of important limitations exist in the bioconcentration and bioaccumulation data for endosulfan with aquatic invertebrates.

First, except for the 122-d study by Roberts (1972) with blue mussel, the existence of steady-state conditions was not documented in any of the studies. This is potentially important since these studies used exposure durations that were relatively short (10 days or less). The study by Ernst (1977) assumed that steady-state conditions occurred by 7 days with the blue mussel based on the plateau of water concentrations. As explained in Section III, this assumption involves significant uncertainty. Second, the stability of exposure concentrations was either not verified or exposure concentrations dropped substantially over time in nearly all of these studies. Lastly, the quantification of total endosulfan in tissue was inconsistent, with most of the later studies (*i.e.*, >1990) quantifying both isomers and the sulfate degradate, while most of the earlier studies either reported only the parent compounds or did not report which endosulfan constituents were measured (Schimmel *et al*., 1977 being an exception).

C. Overall Conclusions From Empirical Bioconcentration and Bioaccumulation Studies

Based on this preliminary review of endosulfan bioconcentration and bioaccumulation studies, it appears that BCF values for fish are typically around 3000 or less (L/kg w.w.) with the exception of yellow tetra, with a ratio-based BCF of 5,670 (calculated by this reviewer) and a reported kinetic-based BCF of about 11,600. As discussed previously and in Section III, the kinetic based BCF for yellow tetra appears inconsistent with the observed accumulation pattern reported in this study and therefore, should be considered with caution. Based only on the two highest quality studies and using the evaluation criteria discussed previously, bioconcentration of endosulfan in fish appears to be in the 1000 to 3000 range (Hansen and Cripe, 1991 for sheepshead minnow and Schimmel *et al*., 1977 for striped mullet, Table A-1).

Depuration of endosulfan (and its metabolite, endosulfan sulfate) by fish appears to be relatively rapid (half life of 2-6 days) which appears inconsistent with some of the observed accumulation patterns measured in these studies.

Bioconcentration of endosulfan by invertebrates appears lower than that reported for fish (600 or less) and data quality is a concern with most of these studies. One study directly evaluated the importance of food vs. water uptake of endosulfan (DeLorenzo *et al* 2002), which suggests that uptake from food (i.e., trophic transfer) by zooplankton (*D. magna*) is minor relative to uptake from water (bioconcentration).

Table A-1. Summary of Aquatic Bioconcentration Studies with Endosulfan

Chemical (formulation/ % ai) (*1)	Species	Study Design (*2)	Exposure Duration (Exposure Conc. µg/L)	BCF Method (SS) (*3)	Avg. BCF/ (BAF)	Range [SD] BCF/ (BAF)	N	Reference
Endosulfan 64% α / 36% β (TG/ 98%)	Sheepshead minnow *(Cyprinodon variegatus)*	FT / M / WB	28 d (5 levels, ~0.05-5.5)	Ratio, α+ β (SS NR)	1146 (*4)	318-2963	9	Hansen & Cripe (1991)
Endosulfan 2:1 α / β (TG/97%)	Zebra Fish *(Brachydanio rerio)*	SR / U / WB	21 d (1 level, 0.3)	Kinetic, α+ β+ sulfate	2650	[441]	3	Toledo and Jonsson (1992)
Endosulfan 2:1 α / β (TG/97%)	Yellow Tetra *(Hyphessobrycon bifasciatus)*	SR / U / WB	21 d (1 level, 0.3)	Kinetic, α+β+ sulfate Ratio	11583(*5) 5670	[2361] ---	3 3	Jonsson and Toledo (1993)
endosulfan + 6 organochlorine pesticides (NR)	Blue Mussel *(Mytilus edulis)*	S / M / WB	7 d (1 level, 2.1➔0.14)	Ratio (SS assumed)	600	NR	NR	Ernst (1977) (*6)
Endosulfan 70% α / 30% β (TG, ai NR)	Striped mullet *(Mugil cephalus)*	FT / M / WB	28-d (1 level, 0.035 ± 0.006)	Ratio, α+ β+ sulfate (non-SS?)	2,755	NR	5	Schimmel *et al* (1977) (*6)
	Striped Mullet *(Mugil cephalus)*	FT / M / WB	96-h (3 levels, 0.36-0.49)	Ratio, α+ β+ sulfate (non-SS)	1115	1000-1344	3	
	Spot *(Leiostomus xanthurus)*	FT / M / WB	96-h (3 levels, 0.05-0.31)	Ratio, α+ β+ sulfate (SS NR)	780	620-895	3	
	Grass shrimp *(Palaemonetes pugio)*	FT / M / WB	96-h (5 levels, 0.16-1.75)	Ratio, α+ β+ sulfate (SS NR)	175	81-245	5	
	Pinfish (*Lagodon rhomboids)*	FT / M / WB	96-h (2 levels, 0.15-0.26)	Ratio, α+ β+ sulfate (SS NR)	1173	1046-1299	2	
Endosulfan (NR)	Blue Mussel *(Mytilus edulis)*	FT / U / WB	122-d (3 levels, 100-1000)	Ratio, α+ β (non-SS?)	12	8-17	3	Roberts (1972)
Endosulfan (NR)	Striped mullet *(Mugil cephalus)*	FT / M / Muscle	10-d (3 levels, 0.13- 1.25)	Ratio (SS NR)	18.4	18.1-18.6	3	Rajendran and Venugopalan (1991)
	Catfish *(Mystus gulio)*	FT / M / Muscle	10-d (3 levels, 0.2- 1.95)	Ratio (SS NR)	17.1	16.6-17.5	3	

Chemical (formulation/ % ai) [*1]	Species	Study Design [*2]	Exposure Duration (Exposure Conc. µg/L)	BCF Method (SS) [*3]	Avg. BCF/ (BAF)	Range [SD] BCF/ (BAF)	N	Reference
	Oyster *(Crassostrea madrasensis)*	FT / M / Foot	10-d (3 levels, 0.14- 1.41)	Ratio (SS NR)	60	42-70	3	
	Clam *(Katelysia opima)*	FT / M / Foot	10-d (3 levels, 0.14- 1.41)	Ratio (SS NR)	46	30-61	3	
Endosulfan (NR)	Crayfish *(Procambarus clarkii)*	NR / U / WB	56-d (100)	Ratio, , α+ β+ sulfate (non-SS)		≤ 1.9[*7]	---	Naqvi and Newton (1990)

[*1] TG = technical grade; ai = active ingredient; NR = not reported.
[*2] FT = flow through; R = static renewal; S = static; M = measured exposure conc.; U = unmeasured exposure conc. WB = whole body, M=muscle, F=foot
[*3] Ratio method = ratio of tissue to water concentration; Kinetic method = ratio of uptake to elimination rate; SS = steady state. All BCFs are expressed on a wet weight basis.
[*4] Average BCFs reported here are calculated from 9 acceptable tests reported by the authors and from treatments with no statistically significant effects on survival or growth relative to controls.
[*5] Kinetic-based BCF is questionable because elimination half-life derived from K2 is not consistent with observed data. A 21-d BCF (ratio method) of 5670 is calculated based on total endosulfan (α, β, sulfate).
[*6] BCF data included in EPA's 2002 Ecological Risk Assessment.
[*7] BCF value from this study is highly suspect due to irregular accumulation patterns and study design issues.

Table A-2. Summary of Aquatic Bioaccumulation Studies with Endosulfan

Species	Study Location/ Design	Analytes	Water Conc. (µg/L)	Sediment Conc. (µg/kg)	Tissue Conc. (ug/kg w.w)	BAF [BSAF]	N	Reference
Mussel *(Mytilus galloprovincialis)*	Black Sea (4 coastal stations)	Endosulfan sulfate	<0.01	< 0.01-25	<0.01-0.08	[0.059]	4	Ozkoc and Bakan, 2007
Oyster *(Crassostrea virginica)*	Mesocosm (96-h, 70:30 α:β)	Total endosulfan (α+β+sulfate)	3 levels; 0.18➔0.06 0.52➔0.12 3.0➔0.29	ND (< 32)	35-606	637 ± 189	3	Pennington *et al* (2004)
Green alga *(Pseudokirch-neriella subcapitatum)*	Microcosm (24-h TG 2:1 α:β)	Total endosulfan (α+β+sulfate)	100	NA	53.6 (*1)	536(*1)	---	DeLorenzo *et al* (2002)
Water flea *(Daphnia magna)*	Microcosm (24-h TG 2:1 α:β)	Total endosulfan (α+β+sulfate)	100 (*2) 100(*2)+food food only	NA	65.6(*1) 62.4(*1) 1.68(*1)	656(*1) 624(*1) 16.8(*1)	---	DeLorenzo *et al* (2002)

(*1) Tissue concentrations and BCF converted from dry wt to wet wt. assuming 80% water fraction in tissue.

(*2) Water concentrations based on nominal values.

II. INDIVIDUAL STUDY SUMMARIES: BIOCONCENTRATION

Hansen and Cripe, 1991 (Sheepshead Minnow)

Summary. An interlaboratory comparison of the early life stage toxicity of endosulfan (technical grade, 98% ai) to the sheepshead minnow was conducted by Hansen and Cripe (1991). Although this study was not designed to assess bioconcentration *per se*, endosulfan residues were measured in fish at test termination (day 28). Of the 14 endosulfan tests conducted by 7 laboratories, 9 were considered acceptable by the authors based on control survival, variability in exposure concentrations and adherence to other ASTM protocols. Continuous flow-through endosulfan exposures began with embryos and continued through 28 days. Concentrations of endosulfan (α+ β) were measured in exposure chambers and in fish from 5 treatments and two controls (negative and unspecified solvent control). In accordance with ASTM and OPP guidelines on bioconcentration studies, BCFs reported in Table A-1 were calculated only from treatments without significant effects on survival and growth relative to controls (*i.e.*, organism stress can alter accumulation kinetics and BCFs). Using data from treatments without adverse effects, the mean BCF across all 9 acceptable tests was 1146 (L/kg w.w.) and ranged by about a factor of 10 across laboratories (approximately 300 to 3000). Average BCFs reported by the authors (which included data from unacceptable tests and treatments with adverse effects) ranged from 350 to 3700 (overall mean: 1300). These BCFs are similar to the BCFs reported in Table A-1 from acceptable tests and treatments, suggesting that organism stress did not substantially impact BCF values.

Data Quality/Interpretation. Several limitations of this study should be considered when interpreting these bioconcentration results. First, the existence of steady-state conditions could not be confirmed since only one measurement of endosulfan residues was made at test termination. However, if biological half lives in larval sheepshead minnow are similar to those reported for other adult fish (on the order of a few days), steady-state conditions would have been reached in the study. Second, the BCFs reported are based on parent compound only (endosulfan isomers) and do not include the primary degradate (endosulfan sulfate) which is consider of similar toxicity as the parent compound. To the extent that the sulfate degradate was formed by larval fish, the BCFs reported would underestimate the total residue accumulation of endosulfan and its primary metabolites. Lastly, the fish used in the study were by design, actively growing throughout the exposure period. Thus, the phenomenon known as 'growth dilution' could have occurred thereby reducing the magnitude of BCF values compared to non-actively growing fish. Aside from these limitations, this study has a number of strengths including the use of flow-through conditions, rigorous QA on the analytical chemistry, and measured concentrations in water with acceptable temporal variability.

Toledo and Jonsson, 1992 (Zebra Fish)

Summary. The bioaccumulation and elimination of endosulfan (α, β, and endosulfan sulfate) was studied in zebra fish, *(Brachydanio rerio)* for 21 days. In this study, Toledo

and Jonsson (1992) exposed six replicates of adult zebra fish to a single nominal concentration of 0.4 µg/L technical grade endosulfan (2:1 α:β isomeric ratio). Residues were analyzed from three replicates on day 3, 7, 14 and 21. After 21 days, remaining fish were allowed to depurate for an additional 5 days with further measurement of residues. A static-renewal system was used with a 24-h renewal cycle. Exposure concentrations were not measured in the study. Therefore, they were calculated using an estimated first order half-life of 24 hours. BCFs were then calculated using the kinetic method (ratio of uptake rate (K_u) to elimination rate (K_e) for each of the isomers but not endosulfan sulfate (apparently Ku could not be determined). BCFs ranged from approximately 1400 for β-endosulfan to 2650 for total endosulfan (α+β+sulfate). Calculated first order half lives ranged from 2.9 d for α-endosulfan to 5.6 days for endosulfan sulfate. An apparent steady state appeared to be reached within 15-21 days for α, β, and total endosulfan, although endosulfan sulfate residues were still increasing somewhat at 21 days.

Data Quality/Interpretation. The primary limitation in this study relates to the use of a static renew system (rather than a flow through system that is generally required for bioconcentration studies) and subsequent uncertainties associated with the actual exposure encountered by the fish. Exposure concentrations were assumed to drop 50% over each 24-hr renewal period (0.4 to 0.2 µg/L) based on the investigators' previous studies that indicated a 24-h half life for endosulfan in water. Assuming this 50% decline actually occurred, the resulting impact would likely be an underestimation of the uptake rate (K_u) and the steady-state BCF. It is also possible that exposure concentrations dropped even further than 50% due to chemical uptake into organisms (90 fish were initially housed per 17-L replicate). Based on measured total endosulfan residues on day 21 reported in this study (approximately 0.8 µg/g) and estimated water concentration of 0.3 µg/L, a BCF of 2680 can be calculated (ratio method). This value is very close to the kinetic BCF of 2650, suggesting the assumption of first order accumulation kinetics is appropriate for this study.

Jonsson and Toledo, 1993 (Yellow Tetra)

Summary. The bioaccumulation and elimination of endosulfan (α, β, and endosulfan sulfate) was studied in the yellow tetra, *(Hyphessobrycon bifasciatus)* for 21 days. In this study, Jonsson and Toledo (1993) exposed 3 replicates of adult yellow tetra fish to a single nominal concentration of 0.4 µg/L technical grade endosulfan (2:1 α:β isomeric ratio). Residues were analyzed from three replicates on day 3, 7, 14 and 21. After 21 days, remaining fish were allowed to depurate for an additional 5 days with further measurement of residues. A static-renewal system was used with a 24-h renewal cycle. Exposure concentrations were not measured in the study. Therefore, the water concentration was calculated using an estimated first order half-life of 24 hours. BCFs were calculated using the kinetic method for each of the isomers but not endosulfan sulfate because K_u could not be determined. Steady state was not reached in 21 days for α and β endosulfan as residues of these isomers continued to increase over this time period. Residues of endosulfan sulfate were a relatively small fraction of total endosulfan and remained relatively constant over the exposure period after day 7. Kinetic-based BCFs ranged from approximately 9900 for β-endosulfan to 11580 for total endosulfan

(α+β). Calculated first order half lives ranged from 1.7 to 2.0 days for α and β endosulfan, respectively, which is not consistent with the observed data (i.e., using these half-lives, steady state should have been reached within approximately 7 days).

Data Quality/Interpretation. The experimental design used by Jonsson and Toledo (1993) for yellow tetra was identical to that used by the same authors for zebra fish as described previously. Thus, the same limitations apply regarding the use of a use of a static-renewal system and subsequent uncertainties associated with the actual exposure encountered by the fish. Exposure concentrations were assumed to drop 50% over each 24-hr renewal period (0.4 to 0.2 µg/L) based on the investigators' previous studies indicating a 24-h half life for endosulfan in water. Assuming this 50% decline actually occurred, the resulting impact would likely be an underestimation of the uptake rate (K_u) and the steady-state BCF. It is also possible that exposure concentrations dropped even further than 50% due to chemical uptake into organisms (23 fish were initially housed per 17-L replicate). On the other hand, the reported values for depuration rate (K_2) range from 0.34 to 0.39^{-d} and translate into first-order half lives of about 2 days. These half lives appear inconsistent with the measured endosulfan accumulation in fish, which did not appear to reach steady state after 21 days exposure. This inconsistency raises questions whether the assumptions of the kinetic method were satisfied (e.g., first order accumulation kinetics). As an alternative, a measured, non-steady state BCF for total endosulfan is 5670 based on measured concentrations at day 21 and an assumed water concentration of 0.3 µg/L (ratio method).

Schimmel *et al*., 1977 (Striped Mullet, Pinfish, Spot, Grass shrimp)

Summary. Bioconcentration and depuration of endosulfan (70:30 α:β) were studied using a 28-d flow through exposure with juvenile striped mullet (Schimmel *et al*., 1977). Mullet were exposed in duplicate aquaria (n = 100/aquarium) to nominal concentrations of 0.008 and 0.08 µg/L endosulfan. Residues (n=5) were collected over time for analysis in addition to water concentrations. Recovery of spiked residues was 85% (results not corrected). Endosulfan (α, β, sulfate) was not detected in water or tissue at the 0.008 µg/L treatment (DL 0.01 ppb in water, 0.01 ppm in tissue). In the 0.08 µg/L treatment (mean measured concentration of 0.035 µg/L), accumulation was rapid in the first 48 hours, reaching 0.056 ppm total endosulfan (α, β, sulfate) where it remained at or below this level until day 22. On day 22 and 28, tissue concentrations of total endosulfan increased from 0.065 to 0.097 ppm. The vast majority of endosulfan in tissue was present as the metabolite, endosulfan sulfate. The whole body BCF calculated on day 28 was 2755, which might not reflect steady-state conditions. Depuration of endosulfan was rapid, with no endosulfan sulfate measured in mullet tissues after 2 days.

Schimmel *et al* also conducted a series of 96-h flow-through acute toxicity tests of endosulfan with striped mullet, pinfish, spot and grass shrimp. Although not designed to provide steady-state BCF values, residues of endosulfan (α, β, sulfate) were measured in surviving organisms at 96-h. The BCFs for surviving mullet ranged from 1000-1344, which was similar to 96-h measurements in the bioconcentration study, despite substantial mortality and stress in the acute toxicity study. The 96-h BCFs for pinfish,

spot, and grass shrimp were 1046-1299, 620-895, and 81-245, respectively. Based on comparison to the 28-d bioconcentration study, the 96-h BCF for striped mullet does not represent steady-state conditions. It is uncertain whether BCFs for the other species represent steady-state conditions since long-term accumulation data were not available.

Data Quality/Interpretation. The only major limitation identified with the 28-d bioconcentration study with striped mullet is uncertainty in whether the 28-d BCF reflects steady-state conditions. Endosulfan concentrations in edible tissues increased throughout the exposure period while those in whole body increased initially, leveled off, then increased again on day 22 and 28. Increases in tissue concentrations could reflect a reduction in growth rate of juvenile fish but no information was presented regarding growth of organisms. The 28-d bioconcentration study has several strengths, including measurement of chemical concentrations in tissue and water, use of flow-through exposures at sublethal concentrations, and measurement of both endosulfan isomers and its principle degradate (sulfate). The BCFs reported from the 96-h acute toxicity tests are limited primarily by their short exposure period (steady state is uncertain) and the occurrence of severe stress on the surviving organisms (i.e., exposure to lethal concentrations) which could disrupt chemical accumulation kinetics.

Ernst, 1977 (Blue Mussel)

Summary. Ernst (1977) examined the bioconcentration of a mixture of seven organochlorine pesticides (including endosulfan) in the blue mussel, *Mytilus edulis* via 7-d static exposures. Mussels (field collected) were exposed an initial concentration of 2.1 ppb endosulfan (formulation not reported) concurrently with approximately 2 ppb each of α-HCH, γ-HCH, heptachlorepoxide, dieldrin, endrin and DDD. Water concentrations were measured at daily intervals and declined to steady levels by approximately 50 hours for all pesticides. At this point, it was assumed that pesticide concentrations in mussels were at steady state with the water column. The assumed steady-state concentration for endosulfan was 0.14 ppb (exact exposure time not reported). Based on this concentration and the measured concentration of 84 ppb in mussel tissue, a BCF of 600 was reported for endosulfan . Reported endosulfan residues in mussel tissue were adjusted to reflect the % recovery (mean of 56%). Following the 7-d exposure, mussels were transferred to clean water to evaluate pesticide depuration. A half-life of 1.5 d was determined for endosulfan.

Data Quality/Interpretation. Several limitations render the study by Ernst (1977) of questionable value for determining a BCF. First, mussels were exposed to a mixture of pesticides which may have altered accumulation kinetics relative to single chemical exposure. Second, the formulation and composition of endosulfan was not reported for any of the measured concentrations. Third, steady state was assumed based on a plateau of water concentrations over time. This approach assumes that the decline in water concentrations is due entirely to uptake by the organism. However, if the decline in water concentrations was due to degradation of pesticide in water (which occurs based on other laboratory studies), then the assumed steady state concentration in tissue might be

inaccurate. Lastly, endosulfan recoveries in tissue were poor (56%) which adds uncertainty to the quantification of endosulfan tissue residues.

Roberts, 1972 (Blue Mussel)

Summary. The bioconcentration of endosulfan (α, β isomers) by the blue mussel, *Mytilus edulis,* was examined over a 112-d exposure period (Roberts, 1972). In this study, approximately 80 mussels were exposed to nominal concentrations of 0.1, 0.5 and 1.0 mg/L endosulfan plus a control in duplicate 40-L aquaria. Flow-through exposures were maintained using unfiltered seawater. Mussels were not fed during the study. During the first month, mortality occurred in both controls and the 0.1 mg/L treatment, after which they were restocked with fresh mussels. Every two weeks, 6 mussels were removed for chemical analysis of the α and β isomers (but not endosulfan sulfate). Recovery of endosulfan from spiked tissue samples averaged 74% (residues were not corrected for recovery). Accumulation of endosulfan from the 0.1 mg/L nominal concentration remained relatively constant after 50 days, fluctuating between 1.3 and 2.3 mg/kg-w.w. (sum of α & β isomers). Accumulation in the 0.5 and 1.0 mg/L treatments continued to increase over the 112-d exposure period to maxima of 6.5 and 8.1 mg/kg-w.w. total parent compound. Following transfer to clean water, mussels depurated endosulfan in two months to approximately 1-2 mg/kg/w.w. Concentration factors (based on nominal water concentrations) after 112 days exposure ranged from 8 to 17.

Data Quality/Interpretation. The primary limitation of this study is that BCFs are based on nominal water concentrations. To the extent that actual exposures differed from nominal, BCFs would increase or decrease accordingly. Furthermore, mussels were not fed during the study. Analysis of mussel condition index (a growth indicator) indicates a steady decline throughout the study. Thus, mussels may have been held at suboptimal conditions and apparent increases in uptake over the latter portion of the study may actually reflect loss of tissue mass rather than increased uptake. The primary degradate (endosulfan sulfate) was not measured. This study did employ a long exposure period and flow through conditions which are considered strengths.

Rajendran and Venugopalan, 1991 (Striped Mullet, Catfish, Oyster, Clam)

Summary. In 10-d, flow-through exposures, Rajendran and Venugopalan (1991) evaluated the bioconcentration of endosulfan (formulation not reported) in various tissues of striped mullet *(Mugil cephalus)*, long-whiskers catfish *(Mystus gulio),* the Oyster (*Crassostrea madrasensis)*, and the clam, *(Katelysia opima)*. Organisms were housed 35-L fiberglass tanks receiving filtered estuarine water and exposed to three treatments of endosulfan ranging from approximately 0.1 to 2 ppb and control (2 replicates/treatment). Concentrations of endosulfan were measured in water and in tissues. After 10 days, four organisms were sampled for residue analysis. Recovery of spiked endosulfan in tissue was greater than 90%. BCFs for all species and tissues were less than 100. For the fish, average BCFs based on accumulation in muscle were 17-18. For the oyster and clam, BCFs based on accumulation in the foot were 60 and 46, respectively.

Data Quality/Interpretation. In this study, the source, formulation and composition of endosulfan was not reported. Similarly, the isomeric composition associated with the reported concentrations of endosulfan in water and tissue was not described. Therefore, it is uncertain whether or not tissue concentrations reflect one or both isomers or the metabolite, endosulfan sulfate. The existence of steady state is not known, since concentrations were measured just at test termination. If steady state was not reached in 10 days for these organisms, the BCFs would underestimate the steady-state BCFs. The study did use a flow-through exposure system with measured exposure concentrations although temporal variation in measured water concentrations was not reported.

Naqvi and Newton, 1990 (Red Swamp Crayfish)

Summary. The bioconcentration of endosulfan (formulation not reported) was studied in the red swamp crayfish, Procambarus clarkii (Naqvi and Newton, 1990). Crayfish were exposed to nominal concentrations of 100 ppb for 56 days. The authors do not report whether the exposure was static, static-renewal or flow-through. Concentrations of endosulfan (α, β) and endosulfan sulfate were measured from 4 crayfish (2 males and 2 females) at 2, 4 and 8 weeks. Whole body accumulation of α and β endosulfan isomers did not show a time-dependent trend and was highly irregular across individual organisms (i.e., maximum total endosulfan concentration of 200 ppb in one crayfish after two weeks but just 5-16 ppb in the other three samples). At the 4 and 6 week intervals, total endosulfan was generally 10 ppb or lower, except for one sample (72 ppb). Accumulation was not reportedly related to sex of organism. Endosulfan sulfate was detected only in one sample (3 ppb).

Data Quality/Interpretation. Due to the highly irregular accumulation pattern across individual samples observed in this study, and lack of critical information on the study design, it is considered to provide little useful information regarding the accumulation of endosulfan in red swamp crayfish.

III. INDIVIDUAL STUDY SUMMARIES: BIOACCUMULATION

Ozkoc and Bakan, 2007 (Mussel, *Mytilus galloprovincialis*)

Endosulfan sulfate (and other organochlorines) was analyzed in water, sediment and mussels collected from four coastal stations along the Turkish coast of the Black Sea from 2001 to 2003 (Ozkoc and Bakan, 2007). Endosulfan sulfate was detected in mussels at one of the four coastal stations (mean conc. 0.8 ppb) but not in water at any of the stations (DL <0.01 ppb). At this station, the mean concentrations of endosulfan sulfate was 25 ppb. Based on sediment organic carbon and 'extractable organic matter' from mussel tissues (reported as 'mainly lipid'), a biota-sediment accumulation factor of 0.059 was reported by the authors. A bioaccumulation factor could not be calculated because no detectable concentrations were found in water.

Pennington *et al*., 2004 (Oyster, *Crassostrea virginica*)

The acute toxicity and bioaccumulation of endosulfan was evaluated in a mesocosm system by Pennington *et al* (2004). Four endosulfan treatments were evaluated with three replicates per treatment. Mesocosms contained a variety of submerged aquatic vegetation, sediments, and four resident species including pink shrimp, mumichog, fiddler crab, and eastern oyster. Endosulfan residue analysis was only conducted on oyster. In the mesocosm, 12-17 oysters were suspended in the water column and exposed to endosulfan via daily doses of 0.2 ug/L technical grade endosulfan (2:1 α:β isomeric ratio) for 96 hours. Water concentrations were measured daily just after dosing and at the end of the experiment. At test termination (24 hr after the last dose) measured water concentrations of total endosulfan (α+β+sulfate) were less than 5% of the dose-calculated nominal concentrations (0.06 to 0.285 µg/L), and consisted mostly of endosulfan sulfate. Total endosulfan concentrations in oyster tissue ranged from 35 to 600 ppb and consisted mostly of α endosulfan (82%). The mean BAF for oyster based on the time-weighted average water concentration was 637.

DeLorenzo *et al* 2002 (*D. magna, P. subcapitatum*)

The bioaccumulation of endosulfan by *D. magna* from water, water and algae and algae only exposures in laboratory microcosm experiments was evaluated by DeLorenzo *et al* (2002). Algae and daphnids were exposed to 100 ppb nominal concentrations of endosulfan (technical grade, 98% ai) for 16 and 24 hours, respectively. Endosulfan (α, β, and sulfate) measured in the water declined substantially over the exposure period for daphnids (approximating 20% of the nominal concentration). In the daphnid study, endosulfan sulfate represented a relatively small fraction of total endosulfan regardless of exposure route (< 18%). Based on nominal concentrations, BCFs for daphnids were similar between water only and water and food (contaminated algae) exposures (656 and 624, respectively, based on wet weight assuming 80% water fraction in tissue). Uptake of endosulfan from contaminated algae was negligible, suggesting that this exposure route may not be a dominant source of exposure for *D. magna*. Endosulfan in the contaminated algae consisted mostly of the α isomer (77%) and β isomer (14%).

ATTACHMENT B.

PRELIMINARY ASSESSMENT OF BIOACCUMULATION OF ENDOSULFAN AND ASSOCIATED RISKS TO PISCIVOROUS MAMMALS AND BIRDS

Model description and parameterization

The bioaccumulation model used in this assessment relies upon 8 equations for predicting the concentration of endosulfan in tissues of aquatic organisms (Arnot and Gobas 2004). Tissue residues are first calculated at the lowest level of the aquatic food chain (phytoplankton). Concentrations of endosulfan residues are then calculated for zooplankton, including consideration that the diet of zooplankton includes phytoplankton, which contain endosulfan residues. Tissue residues are then calculated for the next 5 trophic levels based on their diets of organisms from lower trophic levels. The equations, their parameters and associated assumptions are described below (**Tables 1-8**). Parameter definitions and abbreviations are consistent with those published by Arnot and Gobas (2004) in order to ensure consistency with the publication and transparent methodology used in this assessment. Ecosystem specific input parameters, such as organism body composition, temperature, and trophic level diets, see **Tables 9-10**.

In order to understand the distribution of possible endosulfan residue tissue concentrations, including mean, standard deviation and 90th percentile values for each aquatic trophic level, parameters were assigned distributions and assumptions of ranges, means and standard deviations. From this, a Monte Carlo simulation was carried out using Crystal Ball 2000. In this simulation, 10,000 trials randomly selected parameters and predicted endosulfan residue concentrations in organisms.

Table 1. Equation A.1, calculation of pesticide tissue residue (C_B) for single trophic levels and its associated parameters and assumptions.

$$Eq.A.1 \quad C_B = \frac{k_1 * (m_0 * \Phi * C_{WTO} + m_P * C_{WDP}) + k_D * \Sigma(P_i * C_{Di})}{k_2 + k_E + k_G + k_M}$$

Parameters:

Symbol	Definition	Value	Units
C_B	pesticide concentration in the organism	calculated	g/kg
C_{BR}	pesticide concentration in the organism originating from uptake through respiration, this parameter is used to calculate BCF	calculated	g/kg
C_{Di}	concentration of pesticide in i (prey item)	calculated	g/kg
C_S	concentration of the chemical in sediment (dry weight of sediment)	Equation A.3	g/(kg (dry) sediment)
C_{WDP}	freely dissolved pesticide concentration in pore water of sediment	input parameter (from PRZM/EXAMS)	g/L

C_{WTO}	total pesticide concentration in water column above the sediment	input parameter (from PRZM/EXAMS)	g/L
k_1	pesticide clearance rate constant through respiratory area (i.e. gills, skin)	Equation A.4	L/kg*d
k_2	rate constant for elimination of the pesticide through the respiratory area (i.e. gills, skin)	Equation A.5	d^{-1}
k_D	pesticide clearance rate constant for uptake through ingestion of food and water	Animals: Equation A.7; Phytoplankton: 0	kg food/ (kg org*day)
k_E	rate constant for elimination of the pesticide through excretion of contaminated feces	Animals: Equation A.8; Phytoplankton: 0	d^{-1}
k_G	organism growth rate constant	Animals: Equation A.6; Phytoplankton: 0.1	d^{-1}
k_M	rate constant for pesticide metabolic transformation	0	d^{-1}
m_o	fraction of resporatory ventilation involving overlying water	1 - m_p	%
m_p	fraction of respiratory ventilation that involves pore-water of sediment	≤5%; 0 for organisms with no contact with pore water	%
P_i	fraction of diet containing i (prey item)	user defined (appendix B)	none
Φ	fraction of the overlying water concentration of the pesticide that is freely dissolved and can be abosrbed via membrane diffusion	Equation A.2	none

Assumptions:

1. The pesticide is distributed homogenously throughout the organism, accounting for phase partitioning.
2. The organism is considered to be a single compartment which exchanges the pesticide with its surrounding environment.
3. Effects of pesticide concentration in egg and sperm tissue are not considered separately.
4. This equation is based on a steady state assumption that involves no change in the concentration of the chemical over time.
5. The growth of an organism over time is constant and can be represented by a constant fraction of the body weight of the organism.
6. Because this assessment considers `endosulfan` as being of concern, and it is possible for the distinct residues to metabolize into other residues of concern, k_M is 0.
7. For plants, k_E is considered insignificant.
8. To calculate C_{BR}, it is assumed that k_D is equal to 0.

Table 2. Equation A.2, derivation of available pesticide fraction in water (Φ) and its associated parameters and assumptions.

$$Eq.A.2 \quad \Phi = \frac{1}{1 + X_{TOC} * K_{OW}}$$

Parameters:

Symbol	Definition	Value	Units
X_{TOC}	concentration of TOC in water	user defined	kg/L
K_{OW}	octanol water partition coefficient	user defined	none
Φ	fraction of the overlying water concentration of the pesticide that is freely dissolved and can be absorbed via membrane diffusion	calculated	none

Assumptions:

1. If a pesticide is associated with organic carbon in the water column, it is not bioavailable to organisms, to the extent that it is in equilibrium with water, some fraction of it is always available.
2. This equation assumes that equilibrium exists between the pesticide concentration in the water and in the organic carbon in the water column.
3. The partitioning of the pesticide into organic carbon in the water column is equal to the phase partitioning into octanol. This is not consistent with Arnot and Gobas (2004), which had separate partition coefficients for POC (0.35) and DOC (0.08). Use of the phase partitioning coefficients increases Φ, resulting in increased available pesticide in the water and increased concentrations of pesticide in organisms (increased C_B values).
4. The sum of particulate organic carbon and dissolved organic carbon is the total organic carbon in the water column.

Table 3. Derivation of pesticide concentration in the solid portion of the sediment (C_S).

$$Eq.A.3 \quad C_S = C_{SOC} * OC$$

$$Where: \quad C_{SOC} = C_{WDP} * K_{OC}$$

Parameters:

Symbol	Definition	Value	Units
C_S	concentration of the chemical in sediment (dry weight of sediment)	calculated	g/(kg (dry) sediment)
C_{SOC}	normalized (for OC content) pesticide concentration in sediment	calculated	g/(kg OC)
C_{WDP}	freely dissolved pesticide concentration in pore water	input parameter (from PRZM/EXAMS)	g/L
K_{OC}	organic carbon partition coefficient	user defined	L/kg OC
OC	percent organic carbon in sediment	user defined	%

Table 4. Equations associated with the derivation of pesticide clearance through the respiratory system (k_1) and associated parameters and assumptions.

$$Eq.A.4.1 \quad ForPhytoplankton: \quad k_1 = \frac{1}{A + B/K_{OW}}$$

$$Eq.A.4.2 \quad ForAnimals: \quad k_1 = \frac{E_W * G_V}{W_B}$$

$$Where: E_W = 1.85 + \left(\frac{155}{K_{OW}}\right)^{-1}$$

$$G_V = 1400 * \left(\frac{W_B^{0.65}}{C_{OX}}\right)$$

$$C_{OX} = (-0.24 * T + 14.04) * S$$

Parameters

Symbol	Definition	Value	Units
A	constant related to the resistance to pesticide uptake through the aqueous phase of plant	6.0×10^{-5} (default)	none
B	constant related to the resistance to pesticide uptake through the organic phase of plant	5.5 (default)	none
C_{ox}	concentration of dissolved oxygen	calculated	(mg O_2)/L
E_W	pesticide uptake efficiency by gills	calculated	%
G_V	ventilation rate of fish, invertebrates, zooplankton	calculated	L/d
k_1	pesticide clearance rate constant through respiratory area (i.e. gills, skin)	calculated	L/kg*d
K_{OW}	octanol water partition coefficient	user defined	none
S	oxygen saturation in water column	user defined	%
T	temperature	user defined	^oC
W_B	wet weight of the organism at t	user defined	kg

Assumptions:

1. The uptake of a chemical by respiratory tissues (E_V) is related to the ventilation rate (G_V) of animals. G_V is related to weight (wet) and oxygen consumption. Uptake efficiency of a pesticide is not measured directly.

2. Rate constants A and B were derived based on empirical data and are not pesticide specific.

3. G_V is based on a single linear relationship for zooplankton, invertebrates and fish.

Table 5. Equations involved with the derivation of the respiratory elimination rate constant (k_2) and associated parameters and assumptions.

$$Eq.A.5 \quad k_2 = \frac{k_1}{k_{BW}}$$

$$Where: \quad k_{BW} = V_{LB} * K_{OW} + V_{NB} * \beta * K_{OW} + V_{WB}$$

Parameters:

Symbol	Definition	Value	Units
k_1	pesticide clearance rate constant through respiratory area (i.e. gills, skin, membrane permeation)	calculated (Equation A.3)	L/kg*d
k_2	rate constant for elimination of the pesticide through the respiratory area (i.e. gills, skin, membrane permeation)	calculated	d^{-1}
k_{BW}	organism-water partition coefficient (based on wet weight)	calculated	none
K_{OW}	octanol water partition coefficient	user defined	none
V_{LB}	lipid fraction of organism	user defined	(kg lipid)/ (kg organism wet weight)
V_{NB}	NLOM (Non Lipid Organic Matter) fraction of animals, NLOC (Non Lipid Organic Carbon) of plants	Phytoplankton: 6.5%; Animals: 20%	kg NLOM/ (kg organism wet weight)
V_{WB}	water content of the organism	user defined	kg water/ (kg organism wet weight)
β	proportionality constant expressing the sorption capacity of NLOM or NLOC to that of octanol	Phytoplankton: 0.35; Animals:0.035	none

Assumptions:

1. k_1 and k_2 are closely related since both parameters involve water ventilation and membrane permeation.

Table 6. Equations involving the derivation of the growth rate constant (k_G) and associated parameters and assumptions.

$$Eq.A.6.1 \quad k_G = 0.0005 * W_B^{-0.2} \quad (T \approx 10\ ^oC)$$

$$Eq.A.6.2 \quad k_G = 0.00251 * W_B^{-0.2} \quad (T \approx 25\ ^oC)$$

Parameters:

Symbol	Definition	Value	Units
k_G	organism growth rate constant	calculated	d^{-1}
T	temperature	user defined	^oC
W_B	wet weight of the organism at t	user defined	kg

Assumptions:

1. If T < 17.5 (midpoint between 10 and 25°C), equation A.6.1 is used. If T > 17.5, equation A.6.2 is used.
2. These equations provide an approximation of growth of aquatic organisms based on weight and temperature. There is some uncertainty associated with these equations, since growth rate can be influenced by additional factors, including species and prey availability.

Table 7. Equations involving the derivation of the pesticide clearance rate constant through diet (k_D) and associated parameters and assumptions.

$$Eq.A.7 \quad k_D = E_D * \frac{G_D}{W_B}$$

$$Where: \quad E_D = (3.0x10^{-7} * K_{OW} + 2.0)^{-1}$$

$$For\ animals\ (except\ fileter\ feeders): \quad G_D = 0.022 * W_B^{0.85} * \exp(0.06 * T)$$

$$For\ fliter\ feeders: \quad G_D = G_V * C_{SS} * \sigma$$

$$G_V = 1400 * \left(\frac{W_B^{0.65}}{C_{OX}} \right)$$

$$C_{OX} = (-0.24 * T + 14.04) * S$$

Parameters:

Symbol	Definition	Value	Units
C_{ox}	concentration of dissolved oxygen	calculated	(mg O_2)/L
C_{SS}	concentration of suspended solids	user defined	kg/L
E_D	dietary pesticide transfer efficiency	calculated	%
G_D	feeding rate of organism	calculated	kg/d
G_V	ventilation rate of gills	calculated	L/d
k_D	pesticide clearance rate constant for uptake through ingestion of food and water	calculated	kg food/(kg org*day)
K_{OW}	octanol water partition coefficient	user defined	none
S	oxygen saturation in water column	user defined	%
T	temperature	user defined	^oC
W_B	wet weight of the organism at t	user defined	kg
σ	efficiency of scavenging of particles absorbed from water	100	%

Assumptions:

1. The equation for G_D applies to fish, zooplankton and invertebrates. The equation was derived based on studies with trout.

2. Empirical E_D values vary from 0-100%. Variability in E_D has been attributed to various factors, including: sorption coefficients of chemicals, composition of diet, digestibility of diet (and more). Based on several different observations, it is assumed that this value can be related to K_{OW}.

3. The equation for E_D is based on a lipid-water (2 phase) resistance model.

4. It is assumed that the scavenging efficiency of filter feeders is 100%.

5. k_D is assumed to be 0 for plants.

Table 8. Equations involving the derivation of the fecal elimination rate constant (k_E) and associated parameters and assumptions.

$$Eq.A.8 \quad k_E = G_F * E_D * \frac{k_{GB}}{W_B}$$

$$Where: \quad E_D = (3.0x10^{-7} * K_{OW} + 2.0)^{-1}$$

$$k_{GB} = \frac{(V_{LG} * K_{OW} + V_{NG} * \beta * K_{OW} + V_{WG})}{(V_{LB} * K_{OW} + V_{NB} * \beta * K_{OW} + V_{WB})}$$

$$V_{LG} = \frac{(1-\varepsilon_L) * V_{LD}}{(1-\varepsilon_L) * V_{LD} + (1-\varepsilon_N) * V_{ND} + (1-\varepsilon_W) * V_{WD}}$$

$$V_{NG} = \frac{(1-\varepsilon_N) * V_{ND}}{(1-\varepsilon_L) * V_{LD} + (1-\varepsilon_N) * V_{ND} + (1-\varepsilon_W) * V_{WD}}$$

$$V_{WG} = \frac{(1-\varepsilon_W) * V_{WD}}{(1-\varepsilon_L) * V_{LD} + (1-\varepsilon_N) * V_{ND} + (1-\varepsilon_W) * V_{WD}}$$

$$G_F = [(1-\varepsilon_L) * V_{LD} + (1-\varepsilon_N) * V_{ND} + (1-\varepsilon_W) * V_{WD}] * G_D$$

$$For\ animals\ (except\ fileter\ feeders): \quad G_D = 0.022 * W_B^{0.85} * \exp(0.06 * T)$$

$$For\ fliter\ feeders: \quad G_D = G_V * C_{SS} * \sigma$$

$$G_V = 1400 * \left(\frac{W_B^{0.65}}{C_{OX}}\right)$$

$$C_{OX} = (-0.24 * T + 14.04) * S$$

Parameters:

Symbol	Definition	Value	Units
C_{ox}	concentration of dissolved oxygen	calculated	(mg O_2)/L
C_{SS}	concentration of suspended solids	user defined	kg/L
E_D	dietary pesticide transfer efficiency	calculated	%
G_D	feeding rate of organism	calculated	kg/d
G_F	egestion rate of fecal matter	calculated	(kg feces)/(kg organism)*d
k_E	rate constant for elimination of the pesticide through excretion of contaminated feces	for animals: calculated for plants: 0	d^{-1}
k_{GB}	partition coefficient of the pesticide between the gastro-intestinal tract and the organism	calculated	none
K_{OW}	octanol water partition coefficient	user defined	none

S	oxygen saturation in water column	user defined	%
T	temperature	user defined	^{o}C
V_{LB}	lipid fraction of organism	user defined	(kg lipid)/ (kg organism wet weight)
V_{LD}	overall lipid content of diet	user defined	kg/kg
V_{LG}	lipid contents in the gut	calculated	(kg lipid)/(kg digesta wet weight)
V_{NB}	NLOM (Non Lipid Organic Matter) fraction of animals, NLOC (Non Lipid Organic Carbon) of plants	user defined	kg NLOM/ (kg organism wet weight)
V_{ND}	overall NLOM content of diet	user defined	kg/kg
V_{NG}	NLOM contents in the gut	calculated	(kg NLOM)/(kg digesta wet weight)
V_{WB}	water content of the organism	user defined	kg water/ (kg organism wet weight)
V_{WD}	overall water content of diet	user defined	kg/kg
V_{WG}	water contents in the gut	calculated	(kg water)/(kg digesta wet weight)
W_B	wet weight of the organism at t	user defined	kg
β	proportionality constant expressing the sorption capacity of NLOM to that of octanol	0.035 for animals	none
ε_L	dietary assimilation rate of lipids	fish: 92%; aquatic inverts: 75%; zooplankton: 72%	%
ε_N	dietary assimilation rate of NLOM	fish: 55%; aquatic inverts: 75%; zooplankton: 72%	%
ε_W	dietary assimilation rate of water	freshwater organisms: 25%	%

Assumptions:

1. G_F is a function of the feeding rate and the digestibility of the diet, which is a function of the composition of the diet.
2. For invertebrates, dietary assimilation efficiencies vary significantly, leading to uncertainty in assigning one value to this parameter.
3. Since hydrophobic chemicals are not likely to be stored in the water of organism tissue, this route is not considered significant to bioaccumulation.

Table 9. Diets of biota of the model ecosystem.						
	% Diet for:					
Organism in diet	Zoo plankton	Benthic Invertebrates	Filter Feeder	Small Forage Fish	Medium Forage Fish	Piscivorous Fish
sediment	0.0%	100.0%	33.0%	0.0%	0.0%	0.0%
phytoplankton	100.0%	0.0%	33.0%	0.0%	0.0%	0.0%
zooplankton	0.0%	0.0%	34.0%	33.0%	33.0%	0.0%
benthic invertebrates	0.0%	0.0%	0.0%	33.0%	33.0%	0.0%
filter feeder	0.0%	0.0%	0.0%	34.0%	34.0%	0.0%
small forage fish	0.0%	0.0%	0.0%	0.0%	0.0%	50.0%
medium forage fish	0.0%	0.0%	0.0%	0.0%	0.0%	50.0%
piscivorous fish	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%

Table 10. Parameters and associated assumptions used for estimating body concentrations, BCF and BAF values of endosulfan.							
Para-meter	**Parameter Description**	**Trophic Level**	**Distribution**	**Mean**	**SD**	**Range**	**Data Source**
A	constant related to the resistance to pesticide uptake through the aqueous phase of plant	Phytoplankton	set value	$6.0x10^{-5}$	N/A	N/A	Arnot and Gobas 2004
B	constant related to the resistance to pesticide uptake through the organic phase of plant	Phytoplankton	set value	5.5	N/A	N/A	Arnot and Gobas 2004
C_{SS}	concentration of suspended solids	All	lognormal	$3.0\ E^{-4}$	$3.0\ E^{-3}$	$1.0\ E^{-6}$ to $2.0\ E^{-1}$	NAWQA 2006
C_{WTO}	total pesticide concentration in water column above the sediment	All	uniform	N/A	N/A	0.1-5.0	PRZM/EXAMS, 60 day values (see **Table 12**)
C_{WDP}	freely dissolved pesticide concentration in pore water	All	uniform	N/A	N/A	0.1-5.0	Assumed to be equivalent to aqueous concentrations.
K_{OC}	Organic carbon partition coefficient	All	lognormal	13600	2600	10000-16000	MRID 41412906
Log K_{OW}	Octanol-water partition coefficient	All	uniform	N/A	N/A	3.55-4.78	**Table 11**
m_p	fraction of respiratory ventilation that involves pore-water of sediment	Phytoplankton	set value	0	N/A	N/A	Arnot and Gobas 2004
		Zooplankton	set value	0	N/A	N/A	
		Benthic Inv.	set value	0.05	N/A	N/A	
		Filter Feeders	set value	0.05	N/A	N/A	
		Sm. Forage Fish	set value	0	N/A	N/A	
		Med. Forage Fish	set value	0	N/A	N/A	
		Piscivores	set value	0	N/A	N/A	
OC	percent organic carbon in sediment	All	lognormal	1.40%	2.50%	0.01-50%	NAWQA 2006
S	oxygen saturation in water column	All	lognormal	83%	33%	0-100%	NAWQA 2006
T	temperature	All	lognormal	14	7.9	0-100	NAWQA 2006
V_{LB}	lipid fraction of organism	Phytoplankton	lognormal	0.50%	0.10%	0.01-1%	Arnot and Gobas 2004
		Zooplankton	lognormal	2.00%	0.20%	0.5-3.5%	Arnot and Gobas 2004
		Benthic Inv.	lognormal	2.00%	0.20%	1.0-3.0%	Arnot and Gobas 2004
		Filter Feeders	lognormal	2.00%	0.20%	1.0-3.0%	Arnot and Gobas 2004
		Sm. Forage Fish	lognormal	6.00%	0.60%	1.0-10.0%	Arnot and Gobas 2004
		Med. Forage Fish	lognormal	6.00%	0.60%	1.0-10.0%	Arnot and Gobas 2004

		Piscivores	lognormal	6.00%	0.60%	1.0-10.0%	Arnot and Gobas 2004
V_{NB}	NLOM (Non Lipid Organic Matter) fraction of animals, NLOC (Non Lipid Organic Carbon) of plants	Phytoplankton	set value	6.50%	N/A	N/A	
		Animals	set value	20%	N/A	N/A	Arnot and Gobas 2004
V_{WB}	water content of the organism	All	N/A	N/A	N/A	N/A	
W_B	wet weight of the organism at t	Phytoplankton	N/A	N/A	N/A	N/A	
		Zooplankton	lognormal	1E-07	1.00E-08	0.00000001 - 0.000001	Arnot and Gobas 2004
		Benthic Inv.	lognormal	0.00001	0.000001	0.000001 - 0.0001	Arnot and Gobas 2004
		Filter Feeders	lognormal	0.0001	0.00001	0.00001 - 0.001	Arnot and Gobas 2004
		Sm. Forage Fish	lognormal	0.01	0.001	0.001-0.1	Arnot and Gobas 2004
		Med. Forage Fish	lognormal	0.1	0.01	0.01-1.0	Arnot and Gobas 2004
		Piscivores	lognormal	1	0.1	0.1-10.0	Arnot and Gobas 2004
X_{TOC}	concentration of TOC in water	All	lognormal	4.43E-06	9.2E-06	0.0000001 - 0.00084	NAWQA 2006
β	proportionality constant expressing the sorption capacity of NLOM or NLOC to that of octanol	Phytoplankton	set value	0.35	N/A	N/A	Arnot and Gobas 2004
		Animals	set value	0.035	N/A	N/A	Arnot and Gobas 2004
ε_L	dietary assimilation rate of lipids	Zooplankton	lognormal	72%	7.20%	55-85%	Arnot and Gobas 2004
		Benthic Inv.	lognormal	75%	7.50%	15-96%	Arnot and Gobas 2004
		Filter Feeders	lognormal	75%	7.50%	15-96%	Arnot and Gobas 2004
		All Fish	lognormal	92%	9%	50-99%	Arnot and Gobas 2004
ε_N	dietary assimilation rate of NLOM	Zooplankton	lognormal	72%	7.20%	55-85%	Arnot and Gobas 2004
		Benthic Inv.	lognormal	75%	7.50%	15-96%	Arnot and Gobas 2004
		Filter Feeders	lognormal	75%	7.50%	15-96%	Arnot and Gobas 2004
		All Fish	lognormal	60%	6%	40-80%	Arnot and Gobas 2004
ε_W	dietary assimilation rate of water	All	set value	25%	N/A	N/A	Arnot and Gobas 2004
σ	efficiency of scavenging of particles absorbed from water	Filter Feeders	set value	100%	N/A	N/A	Maximum Assumption

Table 11. Log K_{OW} values for endosulfan and endosulfan sulfate.			
Chemical	**Stereochemistry**	**Log K_{OW} values cited in Sangster 2007**	**Estimated Log P ***
Endosulfan	unspecified	2.23	3.50
Endosulfan	alpha	3.55-4.74	3.50
Endosulfan	beta	3.62-4.78	3.50
Endosulfan sulfate	N/A	3.66	3.64
* By K_{OW}Win			

Table 12. 60-day average Aqueous EECs (µg/L) of endosulfan generated by PRZM/EXAMS. Reported in table 5 of ERA.		
Crop	**Aqueous EEC (Total Endosulfan)**	**Aqueous EEC (Endosulfan sulfate)**
Apples	0.24	0.13
Cotton	2.5	1.38
Lettuce	1.3	0.69
Pecan	3.8	2.09
Potato	1.6	0.89
Tobacco	1.8	0.97
Tomato	4.9	2.68

Sensitivity Analysis

In order to understand the influence of the input parameters on model predictions of endosulfan residue concentrations in tissue of aquatic organisms, a sensitivity analysis was conducted. This analysis was also carried out as a Monte Carlo analysis. Parameter assumptions were the same as those used in defining the ranges of endosulfan residues in aquatic organism tissues (**Table 10**). The sensitivity of the model to specific parameters was defined by the contribution of each parameter to the variance of the estimation of endosulfan residue concentrations in each of the aqueous trophic levels (**Table 13**). The results of the sensitivity analysis indicate that the bioaccumulation predictions are sensitive to log K_{OW} and endosulfan water concentration. Therefore, the selection of these parameters has the greatest effect on the predictions of the model.

Table 13. Contribution to variance (%) of model parameters to tissue concentrations (C_B) of different trophic levels. Parameters that contributed ≤0.1% variance are not included.							
Parameter	**Phyto plankton**	**Zoo plankton**	**Benthic invert.**	**Filter feeder**	**Small forage fish**	**Med forage fish**	**Piscivorous fish**
Log KOW	**54.2%**	**54.3%**	**59.4%**	**60.4%**	**56.0%**	**57.1%**	**62.8%**
Pesticide conc. in water	**44.7%**	**44.1%**	**38.8%**	**37.1%**	**42.5%**	**41.3%**	**35.5%**
TOC (kg OC/L)	0.7%	0.7%	0.6%	0.5%	0.6%	0.6%	0.5%
filter feeder % lipid	0.0%	0.0%	0.0%	0.7%	0.0%	0.0%	0.0%
small forage fish % lipid	0.0%	0.0%	0.0%	0.0%	0.4%	0.0%	0.0%
benthic invertebrate % lipid	0.0%	0.0%	0.6%	0.0%	0.0%	0.0%	0.0%
concentration of suspended solids (kg/L)	0.0%	0.0%	0.0%	0.6%	0.0%	0.0%	0.0%
zooplankton % lipid	0.0%	0.5%	0.0%	0.0%	0.0%	0.0%	0.0%
Pesticide conc. in pore water	0.0%	0.0%	0.2%	0.2%	0.0%	0.0%	0.0%
piscivorous fish % lipid	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.6%
med. forage fish % lipid	0.0%	0.0%	0.0%	0.0%	0.0%	0.5%	0.0%
% oxygen sat	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%

Bioaccumulation model results

Estimated concentrations of endosulfan in tissue concentrations of organisms in the different trophic levels range from 10^2 to 10^4 µg/kg. Although concentrations (wet weight basis) increase from lower to higher trophic levels (**Table 14**), when expressed on a lipid normalized basis, they do not display increases with increasing trophic level (see Table 2-3 in the body of Appendix 1).

Table 14. Predicted concentrations of endosulfan in aquatic organism tissues (µg/kg) at different trophic levels.					
Trophic Level	**Mean**	**SD**	**25th %**	**75th %**	**90th %**
Endosulfan					
Phytoplankton	1,279	1,290	383	1,739	3,233
Zooplankton	1,280	1,307	376	1,742	3,237
Benthic Invertebrates	1,282	1,271	399	1,749	3,188
Filter Feeders	1,411	1,588	407	1,857	3,476
Small Forage Fish	3,346	3,755	950	4,477	8,461
Medium Forage Fish	3,447	3,684	960	4,648	8,856
Piscivorous Fish	4,682	20,306	1,051	5,860	11,925

Bioconcentration Factors (BCFs) and Bioaccumulation Factors (BAFs) are calculated according to **Equations A9 and A10**, where C_{BR} is the amount of pesticide in the tissue of the organism with respect to intake and excretion through respiratory processes, C_B is the total pesticide concentration in the tissue of the organism taken up through respiration and ingestion, and C_{WTO} is the amount of pesticide present in the water column. Modeled BCF and BAF values for endosulfan are in **Tables 15 and 16**, respectively.

$$Equation\ A9.\quad BCF = \frac{C_{BR}}{C_{WTO}}$$

$$Equation\ A10.\quad BAF = \frac{C_{B}}{C_{WTO}}$$

Table 15. Predicted BCF values of endosulfan at different trophic levels.					
Trophic Level	**Mean**	**SD**	**25th %**	**75th %**	**90th %**
Phytoplankton	499	369	191	729	1,079
Zooplankton	496	370	190	720	1,077
Benthic Invertebrates	525	413	197	761	1,122
Filter Feeders	515	403	196	741	1,102
Small Forage Fish	1,196	887	467	1,737	2,553
Medium Forage Fish	1,184	867	462	1,726	2,527
Piscivorous Fish	1,127	805	457	1,627	2,365

Table 16. Predicted BAF values of endosulfan at different trophic levels.					
Trophic Level	**Mean**	**SD**	**25th %**	**75th %**	**90th %**
Phytoplankton	499	369	191	729	1,079
Zooplankton	500	375	190	726	1,089
Benthic Invertebrates	530	421	199	765	1,132
Filter Feeders	585	577	204	816	1,239
Small Forage Fish	1,308	1,080	477	1,889	2,885
Medium Forage Fish	1,353	1,093	476	1,960	3,049
Piscivorous Fish	1,806	5,439	515	2,511	4,282

Comparison of residues of endosulfan in fish to toxicity data for piscivorous animals

In order to assess risks to mammals and birds consuming aquatic organisms which have bioaccumulated endosulfan, several species were selected, including mink, river otter, belted kingfisher, herring gull, osprey, mallard duck, great blue heron and bald eagle.

Species body weight data (in kg) are consistent with the Wildlife Exposure Factors Handbook (USEPA 1993). Food intake values for mink, herring gull, osprey, great blue heron and bald eagle were taken from data cited in USEPA 1993. Food ingestion rates were estimated for otter (mammal equation) mallard duck and kingfisher (bird equation). Food ingestion rates (FI) were estimated by **Equations A.11 and A.12**, where FI is calculated in kg dry food/kg-bw day and Wt is animal body weight in kg. FI rates were converted from food dry weight/kg-bw day to food wet weight/day by assuming the diet of river otter and belted kingfisher includes food of 75% water by weight. The FI rate for mallard duck was converted from food dry weight/kg-bw day to food wet weight/day by assuming the diet of mallard duck includes food of 80% water by weight (USEPA 1993).

[redacted]

Drinking water intakes (DW) for mammals and birds are calculated based on the **Equations A.13 and A.14** (USEPA 1993); where BW represents the body weight (in kg) of the animal for which the drinking water intake is being assessed. Resulting units of DW are L/day.

$$Eq.A.13 \quad DW = \left(0.099 * BW^{0.09}\right) \quad (mammals)$$

$$Eq.A.14 \quad DW = \left(0.059 * BW^{0.67}\right) \quad (birds)$$

Dose-based (mg/kg-bw day) and dietary-based (ppm or mg/kg-diet day) EECs are estimated assuming that pesticide intake is a function of the amount of pesticide contained in the food and drinking water of an animal. The pesticide concentration in food is based on the concentration of pesticide in the prey items and the percent of each prey item in the diet of the animal. Mink, belted kingfisher, great blue heron, and osprey consume 100% forage fish. River otter, herring gull and bald eagle are assumed to consume 80% forage fish and 20% piscivorous fish. Mallard ducks are assumed to consume 34% phytoplankton, 33% zooplankton and 33% benthic invertebrates (USEPA 1993).

The Dose-based EEC is calculated by **Equation A.15**. The pesticide intake through food is calculated by multiplying the percent of each prey item ($\%_{Prey}$) by the pesticide tissue residue concentration for that prey item (C_{Bprey}). The sum of the pesticide residues ingested through food is converted into units of mg pesticide/kg food. This value is then multiplied by the food intake (in units of kg/kg-bw day) for a resulting value in units of mg pesticide/kg-bw day. The pesticide intake through drinking water is calculated by multiplying the concentration of the pesticide in water (C_{WTO}, which is in units of mg/L) by the water intake (DW, units of L/d) and dividing by the bodyweight. This results in units of mg pesticide/kg-bw day. The sum of pesticide intake through diet and through drinking water is the dose-based EEC.

[redacted]

The Dietary-based EEC is calculated by **Equation A.16**. Pesticide intake through food is calculated by multiplying the percent of each prey item ($\%_{Prey}$) by the pesticide tissue residue concentration for that prey item (C_{Bprey}). The sum of the pesticide residues ingested through food is converted into units of mg pesticide/kg food. This value is then multiplied by the food intake (in units of kg food/kg-bw day) and animal body weight (kg-bw) for a resulting value in units of mg pesticide/day. The pesticide intake through drinking water is calculated by multiplying the bioavailable concentration of the pesticide in water (C_{WTO}) (which is in units of mg/L) by the water intake (DW, units of L/d). This results in units of mg pesticide/day. The sum of pesticide intake through diet and through drinking water is the dietary-based EEC.

[redacted]

Available dose-based toxicity values are adjusted for the weights of the animal tested (e.g. laboratory rat and mallard duck or bobwhite quail) and of the animal for which the risks are being assessed (e.g. mink, bald eagle, etc.). These adjustments are made according to the equations below (USEPA 2006), where: AT = adjusted toxicity value; LD_{50} or NOAEL = endpoint reported by toxicity study; TW = body weight of tested animal (350g rat; 1580g mallard or 178 g Northern bobwhite quail); AW = body weight of assessed animal; x = Mineau scaling factor (default value of 1.15 used) (**Equations A.17 and A.18**).

$$Eq.A.17 \quad AT = \left(LD_{50}\ or\ NOAEL\right)\left(\frac{TW}{AW}\right)^{0.25} \qquad (mammals)$$

$$Eq.A.18 \quad AT = LD_{50}\left(\frac{AW}{TW}\right)^{(x-1)} \qquad (birds)$$

Dose-based EECs are divided by adjusted toxicity values to derive RQ values. Dietary-based EECs are divided by available toxicity values to derive RQ values. RQ values are then compared to Agency levels of concern (LOCs) for non-listed and listed mammals and birds.

Toxicity data for exposures of endosulfan to mammals and birds are available in **Table 17**. The resulting RQs are in **Tables 18 and 19**. The acute risk RQs indicate that residues of endosulfan in fish tissues have the potential to be of concern to some mammals and birds, although the exceedence of Agency's LOCs are relatively modest and occurred for three species at the 90th percentile predictions (Table 18). The chronic risk RQs (dose and diet-based) did not exceed the Agency LOC of 1.0 even at the higher percentiles of model predictions (Table 19).

Table 17. Summary of toxicity of endosulfan to mammals and birds. Bold indicates parameters used for RQ derivation.

Species	Endpoint	Value (ppm)	MRID
Laboratory rat (Rattus norvegicus)	LD_{50}	**10**	0038307
	NOEC*	**15**	00148264
Northern bobwhite quail (Colinus virginianus)	LC_{50}	**805**	22923
	NOEC	60	40261303
Mallard duck (Anas platyrhynchos)	LD_{50}	**28**	136998
	LC_{50}	1053	22923
	NOEC**	**30**	40261302

*Effected endpoint at LOEC: growth
**Effected endpoint at LOEC: growth and reproduction

Table 18. Predicted RQ values for mammals and birds exposed to endosulfan through acute, dose-based exposures. All parameters varied according to Table 10. All dietary-based RQ values for birds are <0.01.					
Organism	**Mean**	**SD**	**25th %**	**75th %**	**90th %**
Dose-Based					
Mink	0.07	0.08	0.02	0.09	**0.18**[1]
River otter	**0.15**[1]	**0.25**[1]	0.04	**0.20**[1]	**0.39**[1]
Belted kingfisher	0.08	0.08	0.02	**0.11**[1]	**0.20**[1]
Herring gull	0.03	0.05	0.01	0.04	0.07
Osprey	0.03	0.03	0.01	0.03	0.07
Mallard duck	0.02	0.02	0.01	0.03	0.05
Great blue heron	0.02	0.02	0.01	0.03	0.05
Bald eagle	0.01	0.02	<0.01	0.02	0.03
[1] **Exceeds LOC (0.1) for acute exposures to listed animals.**					

Table 19. Predicted RQ values for mammals and birds exposed to endosulfan through chronic, dose- and dietary-based exposures. All parameters varied according toTable10.					
Organism	**Mean**	**SD**	**25th %**	**75th %**	**90th %**
Dose-Based					
Mink	0.03	0.03	0.01	0.04	0.08
River otter	0.06	0.11	0.02	0.08	0.16
Dietary-based					
Mink	0.04	0.04	0.01	0.05	0.09
River otter	0.37	0.61	0.10	0.49	0.95
Belted kingfisher	0.01	0.01	<0.01	0.01	0.02
Herring gull	0.03	0.04	0.01	0.04	0.07
Osprey	0.04	0.04	0.01	0.05	0.09
Mallard duck	0.02	0.02	0.01	0.02	0.04
Great blue heron	0.05	0.05	0.01	0.07	0.12
Bald eagle	0.07	0.11	0.02	0.09	0.17

ATTACHMENT C.

SUMMARY OF NEW TOXICITY STUDIES FOR ENDOSULFAN SULFATE

As a result of a 2004 Data Call-in for endosulfan, new data on the primary degrade of endosulfan (endosulfan sulphate) were received and reviewed by the Environmental Fate and Effects Division (EFED). These studies and their classifications are listed in Table C-1.

Table C-1. New Ecological Toxicity Data for Endosulfan Sulfate.

Guideline	Data Requirement (Species Tested)	MRID	Classification	Comment
71-1a 850.2200	Avian Oral Acute (Bobwhite Quail)	46430501	Acceptable	
71-2a 850.2200	Avian Dietary (Bobwhite Quail)	46430502	Supplemental	No definitive endpoint value
71-2b 850.2200	Avian Dietary (Mallard Duck)	46382601	Acceptable	
72-3a 850.1075	Acute Estuarine/ Marine Fish (Sheepshead minnow)	46382603	Supplemental	Test fish wet weight ranged lower than recommended
72-1a 850.1075	Acute Freshwater Fish (Bluegill sunfish)	46382604	Supplemental	Test fish wet weight ranged lower than recommended
Non-Guideline 850.1735	Acute Day Midge Spiked Sediment *(Chironomus tentans)*	46382605	Supplemental	Not designed to fulfill any current guidelines
Non-Guideline 850.1740	Acute Amphipod Spiked Sediment *(Leptocheirus plumulosus)*	46382606	Supplemental	Not designed to fulfill any current guidelines
72-3c 850.1035	Acute Estuarine/Marine Invertebrate *(Americamysis bahia)*	46406401	Acceptable	
Non-Guideline 850.1350	Midge Chronic Toxicity *(Americamysis bahia))*	46781601	Supplemental	Discrepancy in control mortality reported in the study; raw data not included
Non-Guideline 850.1740	Whole Sediment Chronic Toxicity Marine Invertebrates *(Leptocheirus plumulosus)*	46929001	Supplemental	Offspring production was significantly reduced in solvent control as compared to the negative control

I. Terrestrial Endosulfan Sulfate Toxicity Data

Avian Oral Acute (Bobwhite Quail; MRID 464305-01.

The acute oral toxicity of Endosulfan Sulfate (a metabolite of endosulfan) to 18-week old Northern Bobwhite quail (*Colinus virginianus*) was assessed over 14 days. Endosulfan sulfate was administered to the birds via oral gavage at nominal concentrations of 0 (carrier control), 9, 18, 35, 70, 140, and 280 mg a.i./kg bw. Doses were adjusted for percent active ingredient.

Cumulative mortality was 0% at the control, 9, and 18 mg a.i./kg bw levels, 20% at the 35 mg a.i./kg bw level, and 100% at the 70, 140, and 280 mg a.i./kg bw levels. Mortality was swift, occurring within 3 days of administration. The 14-day LD_{50} (with 95% C.I.) was 44 (18-70) mg a.i./kg bw. The NOEL for mortality was 18 mg a.i./kg bw. Hyporeactivity and/or panting were observed in a dose-dependent manner in birds from the ≥35 mg a.i./kg bw dose groups. Surviving birds from the 35 mg a.i./kg bw dose group recovered by day 8. The NOEL for clinical signs of toxicity was 18 mg a.i/kg bw. Analysis of body weight data indicated a statistically-significant difference at the 35 mg a.i./kg bw level at 7 and 14 days following treatment for both males and females. Only the control, 9, 18, and 35 mg a.i./kg bw levels were analyzed due to 100% mortality at the higher test concentrations. The NOEL for body weight was 18 mg a.i./kg bw. Based on visual inspection of the data (lack of replicate data precluded statistical analyses), no apparent treatment-related effect on feed consumption was observed between the control group and the 9, 18, and 35 mg a.i./kg bw levels. The effects observed initially appeared to be aversion. The NOEL for feed consumption was considered to be 35 mg a.i./kg bw. No treatment-related findings were observed at necropsy (of 25%) of the decedent birds.

This toxicity study is scientifically sound and fulfills the guideline requirements for an acute toxicity study using the Northern Bobwhite quail (§71-1a). This study is classified as ACCEPTABLE.

Avian Dietary Acute (Bobwhite Quail; MRID 464305-02)

The acute dietary toxicity of Endosulfan Sulfate (a metabolite of Endosulfan) to 13-day-old Northern Bobwhite quail (*Colinus virginianus*) was assessed over 8 days (5 days with treated feed and 3 day recovery period). Endosulfan Sulfate was administered to the birds in the diet at nominal concentrations of 0 (negative control), 200, 400, 800, 1600, and 3200 ppm. Mean-measured concentrations were <48.3 (LOQ, control), 184, 367, 839, 1857, and 3528 ppm a.i., respectively.

Mortality occurred only at the 3528 ppm a.i. level, with 40% mortality between 3 and 5 days. Therefore, the 8-day dietary LC_{50} was >3528 ppm a.i., and an accurate toxicity category could not be assigned. The NOEC for mortality was 1857 ppm a.i.. Hyporeactivity was observed in 50% of birds from the 1857 ppm a.i. level, and hyporeactivity and ataxia were observed in 60% of surviving birds from the 3528 ppm a.i. level. The NOEC for clinical signs of toxicity was 839 ppm a.i.. Analysis of body weight data revealed a treatment-related adverse affect at the ≥839 levels. The percent inhibition for day 5 mean body weight as compared to the control group was

32% at 1857 ppm a.i. and 41% at 3528 ppm a.i.. The percent inhibition for growth during the 5-day exposure period as compared to controls was 47% at 839 ppm a.i., and 100% at both 1857 and 3528 ppm a.i.. The subsequent NOEC for body weight data was 367 ppm a.i.. Based on visual inspection of the data (lack of replicate data precluded statistical analyses), a notable decrease in feed consumption was observed during the exposure period at the 839, 1857, and 3528 ppm a.i. treatment levels, with 26, 49, and 72% inhibition, respectively, compared to the control. The NOEC for feed consumption was 367 ppm a.i..

This toxicity study is scientifically sound. However, since the LC_{50} exceeded the highest concentration tested, an accurate LC_{50} was not determined, and thus an accurate toxicity category was not assigned. This study therefore does not satisfy the guideline requirement for an avian dietary study with the Northern Bobwhite Quail (§71-2a), and is classified as SUPPLEMENTAL.

Avian Dietary Acute (Mallard Duck; MRID 463826-01)

The acute dietary toxicity of Endosulfan Sulfate (a metabolite of Endosulfan) to 10-day-old Mallard duck (*Anas platyrhynchos*) was assessed over 8 days (5 days with treated feed and 3 day recovery period). Endosulfan Sulfate was administered to the birds in the diet at nominal concentrations of 0 (negative control), 156, 313, 625, 1250, and 2500 ppm a.i.. Mean-measured concentrations were <48.3 (LOQ, control), 170, 385, 644, 1394, and 2891 ppm a.i., respectively.

Cumulative mortality was 10, 40, and 80% at the 644, 1394, and 2891 ppm a.i. treatment levels, respectively. No mortality occurred in the control or ≤385 ppm a.i. treatment levels. The 8-day dietary LC_{50} (with 95% C.I.) was 1642 (1162-2624) ppm a.i., which categorizes Endosulfan Sulfate (a metabolite of Endosulfan) as slightly toxic to Mallard duck on an acute dietary basis. The NOEC for mortality was 385 ppm a.i.. Dose-dependent clinical signs of toxicity were observed at the ≥385 ppm a.i. treatment levels, and included head shaking/muscle twitching, ataxia, and/or hyporeactivity. Effects subsided from all surviving birds by day 7. The NOEC for clinical signs of toxicity was 170 ppm a.i..

Analysis of body weight data revealed a statistically-significant reduction at all treatment levels relative to the control during the exposure period (day 5 body weights and days 0-5 growth analyses). The study authors reported that since all birds increased in body weight during the recovery period, the loss in body weight may have been attributed to avoidance issues. Since body weight for day 8 and growth from days 0-8 resulted in statistically-significant differences relative to the control for the 1394 and 2891 ppm a.i. treatment levels, the reported NOEC for body weight data was 170 ppm a.i..

Based on visual inspection of the data (lack of replicate data precluded statistical analyses), a treatment-related effect on feed consumption was observed at the ≥385 ppm a.i. levels. During the exposure period, the percent inhibition relative to the control group was 25, 35, 62, and 75% at the 385, 644, 1394, and 2892 ppm a.i. treatment levels, respectively. No differences were observed in feed consumption during the recovery period. The subsequent NOEC for feed consumption was 170 ppm a.i.

This toxicity study is scientifically sound and fulfills the guideline requirements for an acute dietary toxicity study using the Mallard duck (§71-2b). This study is classified as ACCEPTABLE.

II. Aquatic Water Column Endosulfan Sulfate Toxicity Data

Estuarine/Marine Fish Acute Toxicity (Sheepshead minnow; MRID 463826-03)

The 96-hour acute toxicity of Endosulfan Sulfate (a metabolite of Endosulfan) to Sheepshead minnow (*Cyprinodon variegatus*) was studied under flow-through conditions. Fish were exposed to Endosulfan Sulfate at nominal concentrations of 0 [negative and solvent (acetone) controls], 0.56, 1.1, 2.3, 4.5, and 9.0 ppb a.i. Mean-measured concentrations were <0.043 (<LOQ, controls), 0.71, 1.2, 2.3, 4.3, and 8.1 ppb a.i., respectively.

After 96 hours of exposure, cumulative mortality was 0% in the negative and solvent controls and in the mean-measured 0.71, and 1.2 ppb a.i. treatment groups, and 15, 85 and 100% in the mean-measured 2.3, 4.3 and 8.1 ppb a.i. treatment groups, respectively. Sub-lethal effects were observed in fish from the ≥2.3 ppb a.i. treatment levels. Effects included a complete loss of equilibrium at the 2.3 ppb a.i. level, and erratic swimming and/or lethargy at the 4.3 and 8.1ppb a.i. levels by 96-hours. The 96-hour LC50 (with 95% C.I.) was 3.1 (2.7-3.7) ppb a.i., which categorizes Endosulfan Sulfate (a metabolite of endosulfan) as very highly toxic to Sheepshead minnow (*C. variegatus*) on an acute toxicity basis. The NOEC and LOEC based on mortality and sub-lethal effects were 1.2 and 2.3 ppb a.i., respectively.

This study is scientifically sound but fails to satisfy the guideline requirements for an acute toxicity study with an estuarine/marine fish, the Sheepshead minnow (§72-3a), because test fish wet-weight ranged (0.23-1.2 g) lower than recommended (0.5-5 g). Consequently, this study is classified as SUPPLEMENTAL. The study provides information that may be useful for future risk assessment purposes.

Freshwater Fish Acute Toxicity (Bluegill Sunfish; MRID 463826-04)

The 96-hour acute toxicity of Endosulfan Sulfate (a metabolite of Endosulfan) to Bluegill Sunfish (Lepomis macrochirus) was studied under flow-through conditions. Fish were exposed to Endosulfan Sulfate at nominal concentrations of 0 [negative and solvent (acetone) controls], 0.38, 0.75, 1.5, 3.0, and 6.0 ppb a.i. Mean-measured concentrations were <0.022 (<LOQ, negative and solvent controls), 0.44, 0.74, 1.5, 2.9, and 6.1 ppb a.i., respectively.

Cumulative mortality was 0% in the negative and solvent control groups, compared to 5, 0, 0, 15, and 100% in the mean-measured 0.44, 0.74, 1.5, 2.9, and 6.1 ppb a.i. treatment groups, respectively. Clinical signs of toxicity were observed in fish from the mean-measured 2.9 and 6.1 ppb a.i. treatment levels. Effects included loss of equilibrium, lying on the bottom of the test vessel, erratic swimming behavior, rapid respiration, and darkened pigmentation. The 96-hour LC50 (with 95% C.I.) was 3.8 (2.9-6.1) ppb a.i., which categorizes Endosulfan Sulfate (a metabolite of endosulfan) as very highly toxic to Bluegill Sunfish (Lepomis macrochirus) on an

acute toxicity basis. The NOEC and LOEC values based on mortality and sub-lethal effects were 1.5 and 2.9 ppb a.i., respectively.

This study is scientifically sound but fails to satisfy the guideline requirements for an acute toxicity study with freshwater fish, warm water species (§72-1a), because test fish weight (0.17-1.2 g) ranged lower than recommended (0.5-5.0 g). Consequently, this study is classified as SUPPLEMENTAL.

Marine/Estuarine Invertebrate Acute Toxicity (*Americamysis bahia;* MRID 464064-01)

The 96-hour acute toxicity of Endosulfan sulfate (a metabolite of Endosulfan) to the saltwater mysid, *Americamysis bahia*, was studied under flow-through conditions. Mysids were exposed to the test material at nominal concentrations of 0 (negative and solvent controls), 1.1, 2.3, 4.5, 9.0, and 18 ppb a.i.; mean measured concentrations were <0.085 (<LOQ; controls), 1.3, 2.2, 4.7, 7.7, and 19 ppb a.i.. Following 96 hours of exposure, no mortalities were observed in the negative and solvent controls and the mean-measured 1.3 and 2.2 ppb a.i. treatment groups. Mortality was 10, 55, and 95% in the mean-measured 4.7, 7.7, and 19 ppb a.i. treatment groups, by 96 hours. The 96-hour LC50 value was 7.9 ppb a.i., which categorizes Endosulfan sulfate as very highly toxic to the saltwater mysid, *A. bahia*, on an acute toxicity basis. Sub-lethal effects observed during the exposure period included surviving mysids that exhibited erratic swimming behavior or lethargy. By 96 hours, several mysids in the 7.7 ppb a.i. treatment group and the one surviving mysid in the 19 ppb a.i. treatment group were lethargic. No sub-lethal effects were observed in the negative and solvent controls and the 1.3, 2.2, and 4.7 ppb a.i. treatment groups. Based on mortality and sublethal effects, the NOEC and LOEC values were 2.2 and 4.7 ppb a.i., respectively.

This study is scientifically valid and fulfills the requirements of an acute LC50 test with an estuarine/marine organism (Subdivision E, §72-3(C) [mysid]). This study is classified as ACCEPTABLE.

Marine/Estuarine Invertebrate Chronic Toxicity (*Americamysis bahia;* MRID 467816-01)

A 28-d flow-through chronic toxicity study of the effects of endosulfan sulfate was conducted with the mysid, *Americamysis bahia*. Mysids were exposed to nominal concentrations of: 0, 0.094, 0.19, 0.38, 0.76, 1.5, and 3.0 µg ai/L. Mean measured concentrations in test solutions were <0.006 (negative and solvent control) , 0.10, 0.18, 0.38, 0.73, 1.4, and 3.0 µg ai/L. Endpoints measured included survival, growth (length and dry weight), and reproduction (offspring/female/day). The NOAEC and LOAEC for survival was determined to be 1.4 and 3.0 µg ai/L, respectively. The reproductive NOAEC and LOAEC were 0.73 and 1.4 µg ai/L respectively. Growth, as determined by total length of males or females was not affected at any treatment concentration (NOAEC of 3.0 µg ai/L and LOAEC of > 3.0 µg ai/L). However, when growth was measured as organism dry weight, male mysids were significantly different from controls at 0.73 µg ai/L (LOAEC) with a NOAEC of 0.38 µg ai/L. Interestingly, the mean dry wt. of females was not affected significantly relatively to the controls at the highest test concentration (3.0 µg ai/L).

This study is considered supplemental because raw data were not provided and there was a discrepancy in the reported survival of control organisms. Control survival was reported as either 77% or 67%, the latter exceeds the guideline recommendation of at least 70%. Furthermore, terminal growth measurements should have been taken for all surviving mysids at study termination. In this study, only surviving paired mysids were measured.

III. Aquatic Sediment Endosulfan Sulfate Toxicity Data

Freshwater Invertebrate Acute Sediment Toxicity (*Chironomus tentans*; MRID 463826-05)

The 10-day acute toxicity of Endosulfan sulfate (a metabolite of Endosulfan) to midge larvae, *Chironomus tentans*, was studied under static-renewal conditions in sediment-spiked exposures (overlying-water was not spiked). Endpoints assessed included survival and growth (dry weight).

The nominal spiked sediment test concentrations were 0 (negative and solvent controls), 0.16, 0.31, 0.63, 1.3, 2.5, and 5.0 ppm a.i. (mg a.i./kg dry sediment). Mean-measured sediment concentrations (Days 0 and 10) were <0.00070 (<LOD; negative and solvent controls) and 0.13, 0.25, 0.56, 1.2, 2.6, and 5.2 ppm a.i., with recoveries of 80, 82, 88, 89, 100, and 100% of the nominal concentrations, respectively. Mean-measured (Days 0 and 10) pore water concentrations were <0.067 (<LOD; negative and solvent controls) and 0.63, 1.7, 2.7, 3.8, 8.5, and 17 ppb a.i., and mean-measured (Days 0 and 10) overlying water concentrations were <0.034 (<LOD; negative and solvent controls) 0.045, 0.17, 0.18, 0.57, 1.2, and 2.2 ppb a.i.

Mean percent survival was 96 and 94, and 96, 79, 68, 59, 56, and 29% for the negative and solvent controls, and the mean-measured 0.63, 1.7, 2.7, 3.8, 8.5, and 17 ppb a.i. pore water concentrations, respectively. The Day-10 NOEC, LOEC, and LC_{50} (with 95% C.I.) for survival was 0.63, 1.7, and 10 (8.8-12) ppb a.i., respectively, based on the mean-measured pore water concentrations. Additionally, the Day-10 NOEC, LOEC, and LC_{50} (with 95% C.I.) for survival was 0.13, 0.25, and 3.1 (2.7-3.5) ppm a.i., respectively, based on the mean-measured sediment treatment concentrations.

Mean dry weight per midge was 1.74 and 1.74, and 1.58, 1.81, 1.92, 1.36, 0.50, and 0.09 mg for the negative and solvent controls, and the mean-measured 0.63, 1.7, 2.7, 3.8, 8.5, and 17 ppb a.i. pore water concentrations, respectively. The Day-10 NOEC, LOEC and EC_{50} (with 95% C.I.) for dry weight was 2.7, 3.8, and 6.4 (5.4-7.5) ppb a.i., respectively, based on the mean-measured pore water treatment concentrations. Additionally, the Day-10 NOEC, LOEC and EC_{50} (with 95% C.I.) for dry weight was 0.56, 1.2, and 1.9 (1.6-2.2) ppm a.i., respectively, based on the mean-measured sediment treatment concentrations. No sub-lethal effects or abnormal behavior was reported for surviving midges in the controls or treatment groups during the exposure period.

This study was designed to fulfill proposed OPPTS Draft Guideline 850.1735 (1996), and does not fulfill any current U.S. EPA FIFRA guideline. This study is classified as SUPPLEMENTAL, and provides information on the 10-Day toxicity of Endosulfan sulfate (a metabolite of Endosulfan) to sediment-dwelling midges (*Chironomus tentans*).

Estuarine/Marine Invertebrate Acute Sediment Toxicity (*Leptocheirus plumulosus*; MRID 463826-06)

The 10-day acute toxicity of Endosulfan sulfate (a metabolite of Endosulfan) to marine amphipods, *Leptocheirus plumulosus*, was studied under static-renewal conditions in sediment-spiked exposures (overlying-water was not spiked). Endpoints assessed included survival and growth (dry weight).

The nominal spiked sediment test concentrations were 0 (negative and solvent controls), 0.50, 1.0, 2.0, 4.0, and 8.0 ppm a.i. (mg a.i./kg dry sediment). Mean-measured sediment concentrations (Days 0 and 10) were <0.0014 (<LOD; negative and solvent controls) and 0.45, 0.86, 1.6, 3.1, and 7.0 ppm a.i., with recoveries of 89, 86, 79, 76, and 87% of the nominal concentrations, respectively. Mean-measured (Days 0 and 10) pore water concentrations were <0.15 (<LOD; negative and solvent controls) and 15, 27, 45, 180, and 250 ppb a.i., and mean-measured (Days 0 and 10) overlying water concentrations were <0.065 (<LOD; negative and solvent controls) 0.21, 0.31, 0.94, 1.2, and 3.6 ppb a.i., respectively.
Mean percent survival was 96 and 98, and 92, 92, 56, 48, and 0% for the negative and solvent controls, and the mean-measured 15, 27, 45, 180, and 250 ppb a.i. pore water concentrations, respectively. The Day-10 NOEC, LOEC, and LC_{50} (with 95% C.I.) for survival was 27, 45 and 73.7 (65.4-83.3) ppb a.i., respectively, based on the mean-measured pore water treatment concentrations. The Day-10 NOEC, LOEC, and LC_{50} (with 95% C.I.) for survival was 0.86, 1.6, and 2.3 (2.1-2.5) ppm a.i., respectively, based on the mean-measured sediment concentrations. No sub-lethal effects were observed throughout the exposure period in the control or treatment groups. Dry weight per amphipod was not assessed for treatment-related reductions in this study.

This study was designed to fulfill proposed OPPTS Draft Guideline 850.1740 (1996), and does not fulfill any current U.S. EPA FIFRA guideline. This study is classified as SUPPLEMENTAL, and provides information on the 10-Day toxicity of Endosulfan sulfate (a metabolite of Endosulfan) to sediment-dwelling marine amphipods (*Leptocheirus plumulosus*).

Estuarine/Marine Invertebrate Chronic Sediment Toxicity (*Leptocheirus plumulosus*; MRID 469290-01)

A 28-d whole sediment chronic toxicity test was conducted with the estuarine amphipod, *Leptocheirus plumulosus,* exposed to endosulfan sulfate. Amphipods were exposed to nominal concentrations of: 0.031, 0.077, 0.19, 0.48, and 1.2 mg ai/kg dw sediment in spiked sediments. All control and treatment levels were analyzed on days 0, 14, and 28 for total [^{14}C]residues using LSC in interstitial water and bulk sediment. Mean measured concentrations in sediments were <0.00086 (controls), 0.032, 0.083, 0.19, 0.47, and 1.2 mg total [^{14}C]endosulfan sulfate equivalents/kg dw sediment (based on LSC analysis). Mean concentrations measure in sediment pore water were <0.24 (solvent and negative controls), 0.23, 0.19, 0.77, 1.58, and 4.0 µg ai/L.

Control mortality was 3% in the negative and solvent controls. For all test levels, survival averaged 89-99%, with no treatment-related differences observed. Dry weight at study

termination and reproduction were the most sensitive endpoints, with statistically-significant reductions from the negative control group at the highest test concentration of 4.0 µg ai/L in pore water (1.2 mg/kg dw in bulk sediment). Survival was not affected at the highest test concentration. A significant difference in reproduction (number of offspring/female) occurred between the negative and solvent control. However, no significant difference in reproduction occurred in all but the highest treatment relative to the negative controls. Since solvent was used in all treatments and the solvent control, this suggests the potential effect of the solvent on reproduction relative to the negative control is not consistently expressed. Although the study could be invalidated on the basis of the reproduction effects in the solvent control, it is considered supplemental because both survival and growth showed no significant differences occurred between solvent and negative controls. The sediment pore water NOAEC and LOAEC for growth (dry weight) are 1.58 and 4.0 µg ai/L, respectively. The sediment pore water NOAEC and LOAEC for survival 4.0 and >4.0 µg ai/L, respectively.

Endosulfan Sulfate Toxicity Study Citations

MRID 464305-01. Stoughton, T. 2004. Endosulfan Sulfate Technical: An Acute Dietary LD50 with Northern Bobwhite. Unpublished study performed by Bayer CropScience, Stilwell, KS. Laboratory ID No. ES 711701. Study sponsored by the Endosulfan Task Force (ETF), West Chester, PA.

MRID 464305-02. Sabbert, T.J. 2004. Endosulfan Sulfate Technical: A Subacute Dietary LC50 with Northern Bobwhite. Unpublished study performed by Bayer CropScience, Stilwell, KS. Laboratory ID No. ES 721701. Study sponsored by the Endosulfan Task Force (ETF), West Chester, PA.

MRID 463826-01. Christ, M.T., and C.V. Lam. 2004. Technical Endosulfan-Sulfate (a Metabolite of Endosulfan): A Subacute Dietary LC_{50} with Mallards. Unpublished study performed by Bayer CropScience, Stilwell, KS. Laboratory ID No. ES720801. Study sponsored by the Endosulfan Task Force (ETF), West Chester, PA.

MRID 463826-03. Cafarella, M.A. 2003. Endosulfan Sulfate - Acute Toxicity to Sheepshead Minnow (*Cyprinodon variegatus*) Under Flow-Through Conditions. Unpublished study performed by Springborn Smithers Laboratories, Inc., Wareham, MA. Laboratory Study No. 13798.6138. Study sponsored by Bayer Crop Science, Research Triangle Park, North Carolina.

MRID 463826-04. Cafarella, M.A. 2003. Endosulfan Sulfate - Acute Toxicity to Bluegill Sunfish (*Lepomis macrochirus*) Under Flow-Through Conditions. Unpublished study performed by Springborn Smithers Laboratories, Inc., Wareham, MA. Laboratory Study No. 13798.6127. Study sponsored by Bayer Crop Science, Research Triangle Park, North Carolina.

MRID 463826-05 Putt, A.E. 2004. Endosulfan Sulfate-Toxicity to Midge (Chironomus tentans) During a 10-Day Sediment Exposure. Study conducted by Springborn Smithers Laboratories, Wareham, Massachusetts. Study sponsored by Bayer CropScience, Research Triangle Park, NC

MRID 463826-06 Putt, A.E. 2004. Endosulfan Sulfate-Toxicity to Marine Amphipods (Leptocheirus plumulosus) During a 10-Day Sediment Exposure. Study conducted by Springborn Smithers Laboratories, Wareham, Massachusetts. Study sponsored by Bayer CropScience, Research Triangle Park, NC

MRID 464064-01 Cafarella, Mark, A. 2003. Endosulfan Sulfate-Acute Toxicity to Mysids (Americamysis bahia) Under Flow-Through Conditions. Unpublished study performed by Springborn Smithers Laboratories, Wareham, Massachusetts. Study sponsored by Bayer CropScience Research Triangle Park, North Carolina.
Cafarella, M.A.

MRID 467816-01. Cafarella, M.A. 2006. Endosulfan Sulfate – Life-Cycle Toxicity Test with Mysids (Americamysis bahia). Unpublished study performed by Springborn Smithers Laboratories, Wareham, MA Study Sponsored by Bayer CropScience LP (on behalf of Endosulfan Task Force) Research Triangle Park, NC

MRID 469290-01. Putt, A.E. Endosulfan Sulfate – Toxicity to Estuarine Amphipods (Leptocheirus plumulosus) During a 28-Day Sediment Exposure. Study conducted by Springborn Smithers Laboratories, Wareham, MA Study sponsored by Bayer CropScience LP on behalf of Endosulfan Task Force.

ATTACHMENT D.

stored as Toma09.out
Chemical: Endosulfan
PRZM envi modified Tueday, 29 May 2007 at 12:54:10
EXAMS en modified Thuday, 29 August 2002 at 16:33:30
Metfile: w12 modified Wedday, 3 July 2002 at 10:04:30
Water segment concentrations (ppb)

Year	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
1961	3.67	2.66	1.87	1.32	1.08	0.29
1962	7.98	6.57	4.01	2.41	2.01	1.09
1963	23.44	17.17	8.69	5.40	4.51	2.00
1964	20.69	15.80	8.79	6.86	6.43	3.84
1965	27.84	20.00	12.40	7.87	6.60	4.02
1966	11.41	9.44	6.04	5.21	4.84	4.35
1967	15.63	11.60	6.61	5.01	4.32	3.13
1968	20.77	17.57	10.57	7.25	6.21	3.90
1969	12.64	10.00	7.05	6.17	5.49	3.98
1970	8.97	7.03	5.38	4.16	4.16	3.33
1971	19.32	14.73	7.28	4.51	4.01	2.46
1972	17.37	12.54	6.08	4.49	4.42	3.42
1973	7.20	6.59	4.86	3.89	3.43	2.65
1974	16.20	11.62	6.35	4.36	3.94	2.81
1975	12.11	8.68	6.40	4.10	3.54	2.36
1976	11.17	8.78	4.46	3.95	3.55	2.59
1977	12.49	9.38	5.84	4.60	4.28	3.24
1978	15.48	12.63	7.60	5.43	4.91	3.41
1979	15.50	11.22	7.62	5.56	4.86	3.57
1980	8.60	6.67	4.24	3.60	3.30	3.01
1981	12.87	9.40	5.10	4.11	3.88	2.50
1982	16.74	12.67	9.30	6.29	5.61	4.14
1983	14.65	10.73	6.61	5.37	4.78	4.01
1984	23.44	18.85	8.71	5.41	5.02	3.61
1985	9.35	7.37	5.93	4.02	3.76	3.04
1986	12.49	9.21	5.08	4.25	3.91	2.89
1987	13.46	10.61	6.64	5.55	4.91	3.17
1988	10.66	7.99	4.81	4.31	4.02	3.23
1989	6.72	5.38	3.47	2.91	2.50	2.09
1990	11.90	9.28	6.74	4.39	3.73	2.40

Sorted results

Prob.	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
0.03	27.84	20.00	12.40	7.87	6.60	4.35
0.06	23.44	18.85	10.57	7.25	6.43	4.14
0.10	23.44	17.57	9.30	6.86	6.21	4.02
0.13	20.77	17.17	8.79	6.29	5.61	4.01
0.16	20.69	15.80	8.71	6.17	5.49	3.98
0.19	19.32	14.73	8.69	5.56	5.02	3.90
0.23	17.37	12.67	7.62	5.55	4.91	3.84
0.26	16.74	12.63	7.60	5.43	4.91	3.61
0.29	16.20	12.54	7.28	5.41	4.86	3.57
0.32	15.63	11.62	7.05	5.40	4.84	3.42
0.35	15.50	11.60	6.74	5.37	4.78	3.41
0.39	15.48	11.22	6.64	5.21	4.51	3.33
0.42	14.65	10.73	6.61	5.01	4.42	3.24
0.45	13.46	10.61	6.61	4.60	4.32	3.23
0.48	12.87	10.00	6.40	4.51	4.28	3.17
0.52	12.64	9.44	6.35	4.49	4.16	3.13
0.55	12.49	9.40	6.08	4.39	4.02	3.04
0.58	12.49	9.38	6.04	4.36	4.01	3.01
0.61	12.11	9.28	5.93	4.31	3.94	2.89
0.65	11.90	9.21	5.84	4.25	3.91	2.81
0.68	11.41	8.78	5.38	4.16	3.88	2.65
0.71	11.17	8.68	5.10	4.11	3.76	2.59
0.74	10.66	7.99	5.08	4.10	3.73	2.50
0.77	9.35	7.37	4.86	4.02	3.55	2.46
0.81	8.97	7.03	4.81	3.95	3.54	2.40
0.84	8.60	6.67	4.46	3.89	3.43	2.36
0.87	7.98	6.59	4.24	3.60	3.30	2.09
0.90	7.20	6.57	4.01	2.91	2.50	2.00
0.94	6.72	5.38	3.47	2.41	2.01	1.09
0.97	3.67	2.66	1.87	1.32	1.08	0.29
0.10	23.17	17.53	9.25	6.80	6.15	4.01
					Average of	3.0164

Inputs generated by pe5.pl - Novemeber 2006

Data used for this run:
Output File: Toma09
Metfile: w12844.dvf
PRZM scen FLtomatoSTD.txt
EXAMS en pond298.exv
Chemical N Endosulfan

Description	Variable Nan	Value	Units	Comments
Molecular w	mwt	406.9	g/mol	
Henry's Lav	henry		atm-m^3/mol	
Vapor Pres	vapr	7.20E-07	torr	
Solubility	sol	3.3	mg/L	
Kd	Kd		mg/L	
Koc	Koc	10600	mg/L	
Photolysis h	kdp	0	days	Half-life
Aerobic Aq	kbacw	2671.2	days	Halfife
Anaerobic /	kbacs	382	days	Halfife
Aerobic Soi	asm	1335.6	days	Halfife
Hydrolysis:	pH 7	19	days	Half-life
Method:	CAM	2	integer	See PRZM manual
Incorporatic	DEPI		cm	
Application	TAPP	1.12	kg/ha	
Application	APPEFF	0.95	fraction	
Spray Drift	DRFT	0.05	fraction of application rate applied to pond	
Application	Date	15-9	dd/mm or dd/mmm or dd-mm or dd-mmm	
Interval 1	interval	7	days	Set to 0 or delete line for single app.
app. rate 1	apprate		kg/ha	
Interval 2	interval	7	days	Set to 0 or delete line for single app.
app. rate 2	apprate		kg/ha	
Record 17:	FILTRA			
	IPSCND	1		
	UPTKF			
Record 18:	PLVKRT			
	PLDKRT			
	FEXTRC	0		
Flag for Ind	IR	EPA Pond		
Flag for run	RUNOFF	none	none, monthly or total(average of entire run)	

stored as Toma09ben.out
Chemical: Endosulfan
PRZM envi modified Tueday, 29 May 2007 at 12:54:10
EXAMS en modified Thuday, 29 August 2002 at 16:33:30
Metfile: w1: modified Wedday, 3 July 2002 at 10:04:30
Benthic segment concentrations (ppb)

Year	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
1961	0.62	0.62	0.62	0.59	0.56	0.15
1962	1.46	1.46	1.44	1.39	1.34	0.78
1963	2.98	2.97	2.94	2.85	2.75	1.41
1964	4.48	4.48	4.45	4.35	4.16	2.89
1965	4.92	4.92	4.89	4.68	4.25	3.28
1966	4.38	4.38	4.35	4.22	4.20	3.75
1967	3.33	3.32	3.28	3.20	3.08	2.67
1968	4.77	4.77	4.72	4.56	4.41	3.02
1969	4.14	4.13	4.09	3.96	3.84	3.33
1970	3.67	3.65	3.61	3.54	3.52	2.99
1971	3.11	3.11	3.07	2.88	2.58	1.93
1972	3.31	3.30	3.26	3.08	3.02	2.74
1973	2.92	2.90	2.83	2.70	2.58	2.29
1974	2.92	2.92	2.88	2.80	2.73	2.23
1975	2.62	2.60	2.53	2.47	2.41	2.02
1976	2.71	2.71	2.68	2.60	2.55	2.09
1977	3.31	3.31	3.23	3.02	3.03	2.51
1978	3.76	3.75	3.71	3.52	3.26	2.76
1979	3.63	3.63	3.58	3.48	3.40	3.00
1980	3.13	3.11	3.05	3.01	3.00	2.61
1981	2.81	2.80	2.77	2.63	2.57	2.02
1982	4.42	4.42	4.36	4.13	3.85	3.14
1983	4.23	4.23	4.19	4.10	4.02	3.39
1984	4.00	4.00	3.95	3.49	3.26	2.91
1985	3.70	3.68	3.58	3.37	3.22	2.74
1986	3.05	3.03	2.97	2.68	2.53	2.22
1987	3.47	3.47	3.45	3.32	3.19	2.59
1988	3.14	3.13	3.09	3.02	2.97	2.73
1989	2.56	2.54	2.47	2.31	2.20	1.85
1990	2.91	2.91	2.89	2.79	2.68	1.85

Sorted results

Prob.	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
0.03	4.92	4.92	4.89	4.68	4.41	3.75
0.06	4.77	4.77	4.72	4.56	4.25	3.39
0.10	4.48	4.48	4.45	4.35	4.20	3.33
0.13	4.42	4.42	4.36	4.22	4.16	3.28
0.16	4.38	4.38	4.35	4.13	4.02	3.14
0.19	4.23	4.23	4.19	4.10	3.85	3.02
0.23	4.14	4.13	4.09	3.96	3.84	3.00
0.26	4.00	4.00	3.95	3.54	3.52	2.99
0.29	3.76	3.75	3.71	3.52	3.40	2.91
0.32	3.70	3.68	3.61	3.49	3.26	2.89
0.35	3.67	3.65	3.58	3.48	3.26	2.76
0.39	3.63	3.63	3.58	3.37	3.22	2.74
0.42	3.47	3.47	3.45	3.32	3.19	2.74
0.45	3.33	3.32	3.28	3.20	3.08	2.73
0.48	3.31	3.31	3.26	3.08	3.03	2.67
0.52	3.31	3.30	3.23	3.02	3.02	2.61
0.55	3.14	3.13	3.09	3.02	3.00	2.59
0.58	3.13	3.11	3.07	3.01	2.97	2.51
0.61	3.11	3.11	3.05	2.88	2.75	2.29
0.65	3.05	3.03	2.97	2.85	2.73	2.23
0.68	2.98	2.97	2.94	2.80	2.68	2.22
0.71	2.92	2.92	2.89	2.79	2.58	2.09
0.74	2.92	2.91	2.88	2.70	2.58	2.02
0.77	2.91	2.90	2.83	2.68	2.57	2.02
0.81	2.81	2.80	2.77	2.63	2.55	1.93
0.84	2.71	2.71	2.68	2.60	2.53	1.85
0.87	2.62	2.60	2.53	2.47	2.41	1.85
0.90	2.56	2.54	2.47	2.31	2.20	1.41
0.94	1.46	1.46	1.44	1.39	1.34	0.78
0.97	0.62	0.62	0.62	0.59	0.56	0.15
0.10	4.48	4.47	4.44	4.34	4.19	3.32
					Average of	2.46

Inputs generated by pe5.pl - Novemeber 2006

Data used for this run:
Output File: Toma09
Metfile: w12844.dvf
PRZM scer FLtomatoSTD.txt
EXAMS en pond298.exv
Chemical N Endosulfan

Description	Variable Na	Value	Units	Comments
Molecular v	mwt	406.9	g/mol	
Henry's La	henry		atm-m^3/mol	
Vapor Pres	vapr	7.20E-07	torr	
Solubility	sol	3.3	mg/L	
Kd	Kd		mg/L	
Koc	Koc	10600	mg/L	
Photolysis	kdp	0	days	Half-life
Aerobic Aq	kbacw	2671.2	days	Halfife
Anaerobic /	kbacs	382	days	Halfife
Aerobic So	asm	1335.6	days	Halfife
Hydrolysis:	pH 7	19	days	Half-life
Method:	CAM	2	integer	See PRZM manual
Incorporati	DEPI		cm	
Application	TAPP	1.12	kg/ha	
Application	APPEFF	0.95	fraction	
Spray Drift	DRFT	0.05	fraction of application rate applied to pond	
Application	Date	15-9	dd/mm or dd/mmm or dd-mm or dd-mmm	
Interval 1	interval	7	days	Set to 0 or delete line for single app.
app. rate 1	apprate		kg/ha	
Interval 2	interval	7	days	Set to 0 or delete line for single app.
app. rate 2	apprate		kg/ha	
Record 17:	FILTRA			
	IPSCND	1		
	UPTKF			
Record 18:	PLVKRT			
	PLDKRT			
	FEXTRC	0		
Flag for Inc	IR	EPA Pond		
Flag for rur	RUNOFF	none	none, monthly or total(average of entire run)	

stored as CAStraw3.out
Chemical: Endosulfan
PRZM envi modified Friday, 8 June 2007 at 20:04:29
EXAMS en modified Thuday, 29 August 2002 at 16:33:30
Metfile: w2: modified Wedday, 3 July 2002 at 10:04:22
Water segment concentrations (ppb)

Year	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
1961	4.61	3.22	2.18	1.26	1.09	0.56
1962	12.48	9.08	4.40	2.92	2.44	1.63
1963	15.93	11.72	6.07	3.89	3.56	2.10
1964	8.99	7.39	4.38	2.67	2.32	1.42
1965	4.91	3.87	3.03	2.15	1.91	1.28
1966	7.26	5.59	3.59	2.40	2.07	1.20
1967	14.60	12.24	7.45	4.16	3.75	2.06
1968	11.84	9.06	4.53	3.10	2.78	1.63
1969	7.17	5.98	5.02	3.94	3.37	1.96
1970	7.36	5.96	4.73	3.15	2.70	1.57
1971	4.45	3.35	2.61	1.81	1.68	1.03
1972	5.60	4.45	2.43	1.55	1.32	0.95
1973	7.88	6.31	4.32	3.48	2.98	1.68
1974	4.57	3.46	2.77	2.04	1.98	1.24
1975	8.99	6.74	3.81	2.67	2.48	1.45
1976	5.78	3.75	2.49	1.98	1.73	1.05
1977	6.25	4.52	2.73	2.18	2.00	1.18
1978	8.94	6.87	4.16	3.77	3.32	1.97
1979	8.26	6.10	3.98	3.41	3.05	1.85
1980	8.96	7.19	4.94	3.75	3.30	1.94
1981	10.30	8.29	4.95	3.09	2.72	1.61
1982	11.45	8.07	4.46	3.67	3.43	2.25
1983	8.79	7.53	5.57	4.13	3.69	2.19
1984	4.77	3.67	2.92	2.05	1.83	1.16
1985	5.98	4.36	2.74	2.02	1.77	1.01
1986	12.06	9.47	4.86	3.82	3.26	1.82
1987	7.10	5.54	3.17	2.50	2.16	1.28
1988	4.72	3.48	2.92	1.77	1.50	1.01
1989	4.34	3.22	2.46	1.76	1.58	0.97
1990	4.23	3.12	2.50	1.80	1.53	0.90

Sorted results

Prob.	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
0.03	15.93	12.24	7.45	4.16	3.75	2.25
0.06	14.60	11.72	6.07	4.13	3.69	2.19
0.10	12.48	9.47	5.57	3.94	3.56	2.10
0.13	12.06	9.08	5.02	3.89	3.43	2.06
0.16	11.84	9.06	4.95	3.82	3.37	1.97
0.19	11.45	8.29	4.94	3.77	3.32	1.96
0.23	10.30	8.07	4.86	3.75	3.30	1.94
0.26	8.99	7.53	4.73	3.67	3.26	1.85
0.29	8.99	7.39	4.53	3.48	3.05	1.82
0.32	8.96	7.19	4.46	3.41	2.98	1.68
0.35	8.94	6.87	4.40	3.15	2.78	1.63
0.39	8.79	6.74	4.38	3.10	2.72	1.63
0.42	8.26	6.31	4.32	3.09	2.70	1.61
0.45	7.88	6.10	4.16	2.92	2.48	1.57
0.48	7.36	5.98	3.98	2.67	2.44	1.45
0.52	7.26	5.96	3.81	2.67	2.32	1.42
0.55	7.17	5.59	3.59	2.50	2.16	1.28
0.58	7.10	5.54	3.17	2.40	2.07	1.28
0.61	6.25	4.52	3.03	2.18	2.00	1.24
0.65	5.98	4.45	2.92	2.15	1.98	1.20
0.68	5.78	4.36	2.92	2.05	1.91	1.18
0.71	5.60	3.87	2.77	2.04	1.83	1.16
0.74	4.91	3.75	2.74	2.02	1.77	1.05
0.77	4.77	3.67	2.73	1.98	1.73	1.03
0.81	4.72	3.48	2.61	1.81	1.68	1.01
0.84	4.61	3.46	2.50	1.80	1.58	1.01
0.87	4.57	3.35	2.49	1.77	1.53	0.97
0.90	4.45	3.22	2.46	1.76	1.50	0.95
0.94	4.34	3.22	2.43	1.55	1.32	0.90
0.97	4.23	3.12	2.18	1.26	1.09	0.56
0.10	12.44	9.43	5.51	3.94	3.55	2.10
					Average of	1.46

Inputs generated by pe5.pl - Novemeber 2006

Data used for this run:

Output File:	CAStraw3			
Metfile:	w23234.dvf			
PRZM scer	CAStrawberry-noplastic_irrig.txt			
EXAMS en	pond298.exv			
Chemical N	Endosulfan			
Description	Variable Na	Value	Units	Comments
Molecular v	mwt	406.9	g/mol	
Henry's La	henry		atm-m^3/mol	
Vapor Pres	vapr	7.20E-07	torr	
Solubility	sol	3.3	mg/L	
Kd	Kd		mg/L	
Koc	Koc	10600	mg/L	
Photolysis	kdp	0	days	Half-life
Aerobic Aq	kbacw	2671.2	days	Halfife
Anaerobic	kbacs	382	days	Halfife
Aerobic So	asm	1335.6	days	Halfife
Hydrolysis:	pH 7	19	days	Half-life
Method:	CAM	2	integer	See PRZM manual
Incorporati	DEPI		cm	
Application	TAPP	1.12	kg/ha	
Application	APPEFF	0.95	fraction	
Spray Drift	DRFT	0.05	fraction of application rate applied to pond	
Application	Date	15-01	dd/mm or dd/mmm or dd-mm or dd-mmm	
Interval 1	interval	7	days	Set to 0 or delete line for single app.
app. rate 1	apprate		kg/ha	
Interval 2	interval	7	days	Set to 0 or delete line for single app.
app. rate 2	apprate		kg/ha	
Record 17:	FILTRA			
	IPSCND	1		
	UPTKF			
Record 18:	PLVKRT			
	PLDKRT			
	FEXTRC	0		
Flag for Inc	IR	EPA Pond		
Flag for rur	RUNOFF	none	none, monthly or total(average of entire run)	

stored as CAStraw3ben.out
Chemical: Endosulfan
PRZM envi modified Friday, 8 June 2007 at 20:04:29
EXAMS en modified Thuday, 29 August 2002 at 16:33:30
Metfile: w2: modified Wedday, 3 July 2002 at 10:04:22
Benthic segment concentrations (ppb)

Year	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
1961	0.61	0.61	0.61	0.59	0.58	0.43
1962	1.59	1.59	1.58	1.53	1.49	1.19
1963	2.52	2.52	2.49	2.46	2.45	1.86
1964	1.83	1.83	1.81	1.74	1.69	1.26
1965	1.53	1.53	1.51	1.45	1.44	1.13
1966	1.62	1.62	1.61	1.55	1.50	1.09
1967	2.32	2.31	2.30	2.23	2.19	1.69
1968	1.99	1.99	1.97	1.94	1.92	1.43
1969	2.32	2.32	2.30	2.25	2.19	1.63
1970	2.01	2.01	2.00	1.98	1.93	1.42
1971	1.27	1.27	1.27	1.24	1.24	0.95
1972	1.05	1.04	1.03	0.99	0.96	0.79
1973	2.04	2.03	2.02	1.97	1.92	1.42
1974	1.50	1.50	1.48	1.44	1.42	1.11
1975	1.71	1.71	1.70	1.64	1.60	1.23
1976	1.33	1.33	1.31	1.26	1.23	0.93
1977	1.37	1.37	1.35	1.33	1.32	1.00
1978	2.25	2.25	2.23	2.19	2.15	1.64
1979	2.16	2.16	2.13	2.10	2.05	1.55
1980	2.49	2.49	2.47	2.38	2.31	1.72
1981	1.93	1.93	1.91	1.88	1.85	1.39
1982	2.53	2.53	2.51	2.41	2.34	1.86
1983	2.66	2.66	2.64	2.57	2.53	1.92
1984	1.61	1.61	1.60	1.54	1.50	1.11
1985	1.26	1.26	1.25	1.23	1.21	0.89
1986	2.26	2.26	2.24	2.14	2.07	1.48
1987	1.69	1.69	1.67	1.62	1.56	1.15
1988	1.20	1.20	1.19	1.14	1.10	0.86
1989	1.16	1.15	1.15	1.13	1.11	0.86
1990	1.13	1.13	1.12	1.08	1.05	0.79

Sorted results

Prob.	Peak	96 hr	21 Day	60 Day	90 Day	Yearly
0.03	2.66	2.66	2.64	2.57	2.53	1.92
0.06	2.53	2.53	2.51	2.46	2.45	1.86
0.10	2.52	2.52	2.49	2.41	2.34	1.86
0.13	2.49	2.49	2.47	2.38	2.31	1.72
0.16	2.32	2.32	2.30	2.25	2.19	1.69
0.19	2.32	2.31	2.30	2.23	2.19	1.64
0.23	2.26	2.26	2.24	2.19	2.15	1.63
0.26	2.25	2.25	2.23	2.14	2.07	1.55
0.29	2.16	2.16	2.13	2.10	2.05	1.48
0.32	2.04	2.03	2.02	1.98	1.93	1.43
0.35	2.01	2.01	2.00	1.97	1.92	1.42
0.39	1.99	1.99	1.97	1.94	1.92	1.42
0.42	1.93	1.93	1.91	1.88	1.85	1.39
0.45	1.83	1.83	1.81	1.74	1.69	1.26
0.48	1.71	1.71	1.70	1.64	1.60	1.23
0.52	1.69	1.69	1.67	1.62	1.56	1.19
0.55	1.62	1.62	1.61	1.55	1.50	1.15
0.58	1.61	1.61	1.60	1.54	1.50	1.13
0.61	1.59	1.59	1.58	1.53	1.49	1.11
0.65	1.53	1.53	1.51	1.45	1.44	1.11
0.68	1.50	1.50	1.48	1.44	1.42	1.09
0.71	1.37	1.37	1.35	1.33	1.32	1.00
0.74	1.33	1.33	1.31	1.26	1.24	0.95
0.77	1.27	1.27	1.27	1.24	1.23	0.93
0.81	1.26	1.26	1.25	1.23	1.21	0.89
0.84	1.20	1.20	1.19	1.14	1.11	0.86
0.87	1.16	1.15	1.15	1.13	1.10	0.86
0.90	1.13	1.13	1.12	1.08	1.05	0.79
0.94	1.05	1.04	1.03	0.99	0.96	0.79
0.97	0.61	0.61	0.61	0.59	0.58	0.43
0.10	2.52	2.51	2.49	2.41	2.34	1.85
					Average of	1.26

Inputs generated by pe5.pl - Novemeber 2006

Data used for this run:
Output File: CAStraw3
Metfile: w23234.dvf
PRZM scer CAStrawberry-noplastic_irrig.txt
EXAMS en pond298.exv
Chemical N Endosulfan

Description	Variable Na	Value	Units	Comments
Molecular v	mwt	406.9	g/mol	
Henry's La	henry		atm-m^3/mol	
Vapor Pres	vapr	7.20E-07	torr	
Solubility	sol	3.3	mg/L	
Kd	Kd		mg/L	
Koc	Koc	10600	mg/L	
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Aerobic Aq	kbacw	2671.2	days	Halfife
Anaerobic ,	kbacs	382	days	Halfife
Aerobic So	asm	1335.6	days	Halfife
Hydrolysis:	pH 7	19	days	Half-life
Method:	CAM	2	integer	See PRZM manual
Incorporati	DEPI		cm	
Application	TAPP	1.12	kg/ha	
Application	APPEFF	0.95	fraction	
Spray Drift	DRFT	0.05	fraction of application rate applied to pond	
Application	Date	15-01	dd/mm or dd/mmm or dd-mm or dd-mmm	
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app. rate 1	apprate		kg/ha	
Interval 2	interval	7	days	Set to 0 or delete line for single app.
app. rate 2	apprate		kg/ha	
Record 17:	FILTRA			
	IPSCND	1		
	UPTKF			
Record 18:	PLVKRT			
	PLDKRT			
	FEXTRC	0		
Flag for Inc	IR	EPA Pond		
Flag for rur	RUNOFF	none	none, monthly or total(average of entire run)	