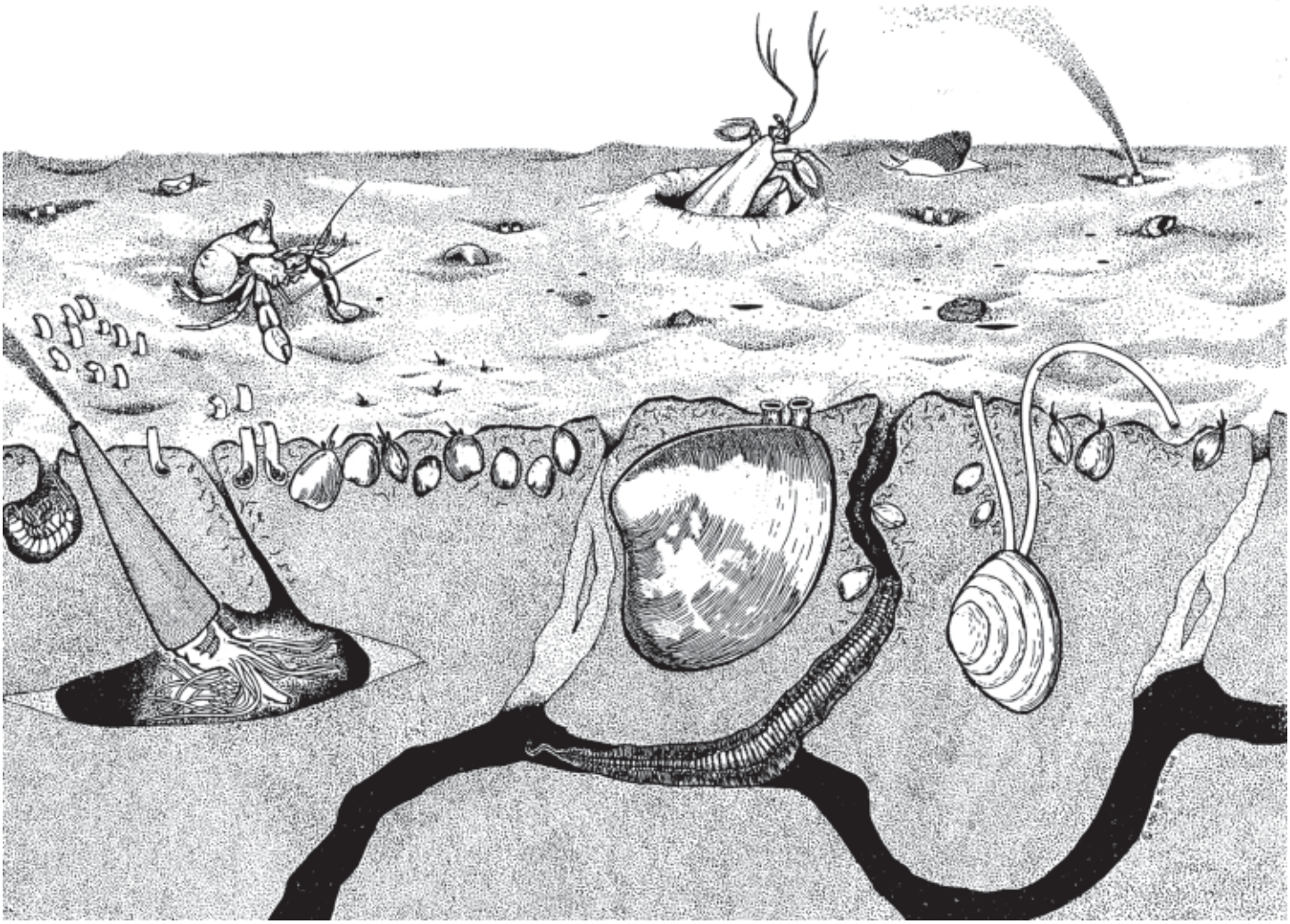
United States
Environmental Protection
Agency

Office of Research and Development
Washington, DC 20460

EPA-600-R-02-009
www.epa.gov

EPA

Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: Endrin



EPA/600/R-02/009
August 2003

Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: Endrin

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Notice

The Office of Research and Development (ORD) has produced this document to provide procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the insecticide endrin. ESBs may be useful as a complement to existing sediment assessment tools. This document should be cited as:

> U.S. EPA. 2003. Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: Endrin. EPA-600-R-02-009. Office of Research and Development. Washington, DC 20460

The information in this document has been funded wholly by the U.S. Environmental Protection Agency. It has been subject to the Agency's peer and administrative review, and it has been approved for publication as an EPA document.

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Abstract

This equilibrium partitioning sediment benchmark (ESB) document describes procedures to derive concentrations of the insecticide endrin in sediment which are protective of the presence of benthic organisms. The equilibrium partitioning (EqP) approach was chosen because it accounts for the varying biological availability of chemicals in different sediments and allows for the incorporation of the appropriate biological effects concentration. This provides for the derivation of benchmarks that are causally linked to the specific chemical, applicable across sediments, and appropriately protective of benthic organisms.

EqP can be used to calculate ESBs for any toxicity endpoint for which there are water-only toxicity data; it is not limited to any single effect endpoint. For the purposes of this document, the Final Chronic Value (FCV) from the Water Quality Criterion (WQC) for endrin was used as the toxicity benchmark. This value is intended to be the concentration of a chemical in water that is protective of the presence of aquatic life. The ESB_{WQC} is derived by multiplying the FCV by the chemical's K_{OC}, yielding the concentration in sediment (normalized to organic carbon) that should provide the same level of protection in sediment that the FCV provides in water. For endrin, this concentration is 5.4 µg endrin/g_{OC} for freshwater sediments and 0.99 µg/g_{OC} for saltwater sediments. Confidence limits of 2.4 to 12 µg/g_{OC} for freshwater sediments and 0.44 to 2.2 µg/g_{OC} for saltwater sediments were calculated using the uncertainty associated with the degree to which toxicity could be predicted by multiplying the K_{OC} and the water-only effects concentration. The ESB_{WQC}s should be interpreted as chemical concentrations below which adverse effects are not expected. At concentrations above the ESB_{WQC}s, effects may occur with increasing severity as the degree of exceedance increases.

The ESBs do not consider the antagonistic, additive or synergistic effects of other sediment contaminants in combination with endrin or the potential for bioaccumulation and trophic transfer of endrin to aquatic life, wildlife or humans.

Foreword

Under the Clean Water Act (CWA), the U.S. Environmental Protection Agency (EPA) and the States develop programs for protecting the chemical, physical, and biological integrity of the nation's waters. To support the scientific and technical foundations of the programs, EPA's Office of Research and Development has conducted efforts to develop and publish equilibrium partitioning sediment benchmarks (ESBs) for some of the 65 toxic pollutants or toxic pollutant categories. Toxic contaminants in bottom sediments of the nation's lakes, rivers, wetlands, and coastal waters create the potential for continued environmental degradation even where water column contaminant levels meet applicable water quality standards. In addition, contaminated sediments can lead to water quality impacts, even when direct discharges to the receiving water have ceased.

The ESBs and associated methodology presented in this document provide a means to estimate the concentrations of a substance that may be present in sediment while still protecting benthic organisms from the effects of that substance. These benchmarks are applicable to a variety of freshwater and marine sediments because they are based on the biologically available concentration of the substance in the sediments. These ESBs are intended to provide protection to benthic organisms from direct toxicity due to this substance. In some cases, the additive toxicity for specific classes of toxicants (e.g., metal mixtures or polycyclic aromatic hydrocarbon mixtures) is addressed. The ESBs do not consider the antagonistic, additive or synergistic effects of other sediment contaminants in combination with endrin or the potential for bioaccumulation and trophic transfer of endrin to aquatic life, wildlife or humans.

ESBs may be useful as a complement to existing sediment assessment tools, to help assess the extent of sediment contamination, to help identify chemicals causing toxicity, and to serve as targets for pollutant loading control measures.

This document provides technical information to EPA Regions, States, the regulated community, and the public. It does not substitute for the CWA or EPA's regulations, nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, States, or the regulated community. EPA and State decisionmakers retain the discretion to adopt approaches on a case-by-case basis that differ from this technical information where appropriate. EPA may change this technical information in the future. This document has been reviewed by EPA's Office of Research and Development (Mid-Continent Ecology Division, Duluth, MN; Atlantic Ecology Division, Narragansett, RI), and approved for publication.

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This is contribution AED-02-046 of the Office of Research and Development National Health and Environmental Effects Research Laboratory's Atlantic Ecology Division.

Front cover image provided by Wayne R. Davis and Virginia Lee.

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Executive Summary

This equilibrium partitioning sediment benchmark (ESB) document describes procedures to derive concentrations of the insecticide endrin in sediment which are protective of the presence of benthic organisms. The equilibrium partitioning (EqP) approach was chosen because it accounts for the varying biological availability of chemicals in different sediments and allows for the incorporation of the appropriate biological effects concentration. This provides for the derivation of benchmarks that are causally linked to the specific chemical, applicable across sediments, and appropriately protective of benthic organisms.

EqP theory holds that a nonionic chemical in sediment partitions between sediment organic carbon, interstitial (pore) water and benthic organisms. At equilibrium, if the concentration in any one phase is known, then the concentrations in the others can be predicted. The ratio of the concentration in water to the concentration in organic carbon is termed the organic carbon partition coefficient (K_{OC}), which is a constant for each chemical. The ESB Technical Basis Document (U.S. EPA, 2003a) demonstrates that biological responses of benthic organisms to nonionic organic chemicals in sediments are different across sediments when the sediment concentrations are expressed on a dry weight basis, but similar when expressed on a μg chemical/g organic carbon basis ($\mu g/g_{OC}$). Similar responses were also observed across sediments when interstitial water concentrations were used to normalize biological availability. The Technical Basis Document further demonstrates that if the effect concentration in water is known, the effect concentration in sediments on a $\mu g/g_{OC}$ basis can be accurately predicted by multiplying the effect concentration in water by the chemical's K_{OC}.

EqP can be used to calculate ESBs for any toxicity endpoint for which there are water-only toxicity data; it is not limited to any single effect endpoint. For the purposes of this document, the Final Chronic Value (FCV) from the Water Quality Criterion (WQC) for endrin was used as the toxicity benchmark. This value is intended to be the concentration of a chemical in water that is protective of the presence of aquatic life. If an FCV is not available, a secondary chronic value (SCV) can be substituted. The ESB_{WQC} is derived by multiplying the FCV by the chemical's K_{OC}, yielding the concentration in sediment (normalized to organic carbon) that should provide the same level of protection in sediment that the FCV provides in water. Ancillary analyses conducted as part of this derivation suggest that the sensitivity of benthic/epibenthic organisms is not significantly different from pelagic organisms; for this reason, the FCV and the resulting ESB_{WQC} should be fully applicable to benthic organisms. For endrin, this concentration is 5.4 μg endrin/g_{OC} for freshwater sediments and 0.99 $\mu g/g_{OC}$ for saltwater sediments. Confidence limits of 2.4 to 12 $\mu g/g_{OC}$ for freshwater sediments and 0.44 to 2.2 $\mu g/g_{OC}$ for saltwater sediments were calculated using the uncertainty associated with the degree to which toxicity could be predicted by multiplying the K_{OC} and the water-only effects concentration. The ESB_{WQC}s should be interpreted as chemical concentrations below which adverse effects are not expected. At concentrations above the ESB_{WQC}s, effects may occur with increasing severity as the degree of exceedance increases. In principle, above the upper confidence limit effects are expected if the chemical is bioavailable as predicted by EqP theory. A sediment-specific site assessment would provide further information on chemical bioavailability and the expectation of toxicity relative to the ESB_{WQC}s and associated uncertainty limits.

As discussed, while this document uses the WQC value, the EqP methodology can be used by environmental managers to derive a benchmark with any desired level of protection, so long as the water-only concentration affording that level of protection is known. Therefore, the resulting benchmark can be species or site-specific if the corresponding water-only information is available. For example, if a certain water-only effects concentration is known to be protective for an

economically important benthic species, the organic carbon-normalized sediment concentration protective for that benthic species could be derived using the effects concentration and the partition coefficient. Such a benchmark might be considered as providing "site-specific protection" for a species or endpoint, if the goal is to derive a benchmark for that particular site or species. Another way to make an ESB site-specific would be to incorporate information on unusual partitioning, if suspected, at the site (see U.S. EPA 2003b).

The ESBs do not consider the antagonistic, additive or synergistic effects of other sediment contaminants in combination with endrin or the potential for bioaccumulation and trophic transfer of endrin to aquatic life, wildlife or humans. Consistent with the recommendations of EPA's Science Advisory Board, publication of these documents does not imply the use of ESBs as stand-alone, pass-fail criteria for all applications; rather, ESB exceedances could be used to trigger the collection of additional assessment data. ESBs apply only to sediments having $\geq$0.2% organic carbon by dry weight.

Tier 1 and Tier 2 ESB values were developed to reflect differing degrees of data availability and uncertainty. Tier 1 ESBs have been derived for endrin in this document, and for the nonionic organic insecticide dieldrin, metal mixtures, and polycyclic aromatic hydrocarbon (PAH) mixtures in U.S. EPA (2003c, d, e). Tier 2 ESBs are reported in U.S. EPA (2003f).

Glossary of Abbreviations

ACR	Acute–chronic ratio
ANOVA	Analysis of variance
AR	Approximate randomization
CFR	Code of Federal Regulations
CWA	Clean Water Act
DOC	Dissolved organic carbon
EC50	Chemical concentration estimated to cause adverse effects to 50% of the test organisms within a specified time period
EPA	United States Environmental Protection Agency
EqP	Equilibrium partitioning
ESB	Equilibrium partitioning sediment benchmark; for nonionic organics, this term usually refers to a value that is organic carbon–normalized (more formally ESB_{OC}) unless otherwise specified
ESB_{OC}	Organic carbon–normalized equilibrium partitioning sediment benchmark
ESB_{WQC}	Equilibrium partitioning sediment benchmark derived based on the Water Quality Criteria for a specific chemical
$ESB_{WQCdry\ wt}$	Dry weight–normalized equilibrium partitioning sediment benchmark derived based on the Water Quality Criteria for a specific chemical
ESB_{WQCOC}	Organic carbon normalized equilibrium partitioning sediment benchmark derived based on the Water Quality Ctieria for a specific chemical
F_1	First progeny generation
FACR	Final acute–chronic ratio
FAV	Final acute value
FCV	Final chronic value
FDA	U.S. Food and Drug Administration
f_{OC}	Fraction of organic carbon in sediment
FRV	Final residue value
GMAV	Genus mean acute value
g_{OC}	Gram organic carbon
HECD	U.S. EPA, Health and Ecological Criteria Division
HMAV	Habitat mean acute value

IUPAC	International Union of Pure and Applied Chemistry
IWTU	Interstitial water toxic unit
K_{OC}	Organic carbon–water partition coefficient
K_{OW}	Octanol–water partition coefficient
K_P	Sediment–water partition coefficient
LC50	The concentration estimated to be lethal to 50% of the test organisms within a specified time period
$LC50_{S,OC}$	Organic carbon–normalized LC50 from sediment exposure
$LC50_W$	LC50 from water-only exposure
NAS	National Academy of Sciences
NERL	U.S. EPA, National Exposure Research Laboratory
NHEERL	U.S. EPA, National Health and Environmental Effects Research Laboratory
NOEC	No observed effect concentration
NTIS	National Technical Information Service
OC	Organic carbon
OEC	Observed effect concentration
OST	U.S. EPA, Office of Science and Technology
PAH	Polycyclic aromatic hydrocarbon
PGMCV	Predicted genus mean chronic value
PSTU	Predicted sediment toxic unit
SCV	Secondary chronic value
SE	Standard error
SMACR	Species mean acute–chronic ratio
STORET	EPA's computerized database for STOrage and RETrieval of water-related data
TOC	Total organic carbon
TU	Toxic unit
WQC	Water quality criteria

Section 1

Introduction

1.1 General Information

Toxic pollutants in bottom sediments of the nation's lakes, rivers, wetlands, estuaries, and marine coastal waters create the potential for continued environmental degradation even where water column concentrations comply with established WQC. In addition, contaminated sediments can be a significant pollutant source that may cause water quality degradation to persist, even when other pollutant sources are stopped. The absence of defensible ESBs make it difficult to accurately assess the extent of the ecological risks of contaminated sediments and to identify, prioritize, and implement appropriate cleanup activities and source controls.

As a result of the need for a procedure to assist regulatory agencies in making decisions concerning contaminated sediment problems, the EPA Office of Science and Technology, Health and Ecological Criteria Division (OST/HECD) and Office of Research and Development National Health and Environmental Effects Research Laboratory (ORD/NHEERL) established a research team to review alternative approaches (Chapman, 1987). All of the approaches reviewed had both strengths and weaknesses, and no single approach was found to be applicable for the derivation of benchmarks in all situations (U.S. EPA, 1989a). The EqP approach was selected for nonionic organic chemicals because it presented the greatest promise for generating defensible, national, numeric chemical-specific benchmarks applicable across a broad range of sediment types. The three principal observations that underlie the EqP approach to establishing sediment benchmarks are as follows:

1. The concentrations of nonionic organic chemicals in sediments, expressed on an organic carbon basis, and in interstitial waters correlate to observed biological effects on sediment-dwelling organisms across a range of sediments.

2. Partitioning models can relate sediment concentrations for nonionic organic chemicals on an organic carbon basis to freely-dissolved concentrations in interstitial water.

3. The distribution of sensitivities of benthic organisms to chemicals is similar to that of water column organisms; thus, the currently established WQC FCV or SCV can be used to define the acceptable effects concentration of a chemical freely-dissolved in interstitial water.

The EqP approach, therefore, assumes that (1) the partitioning of the chemical between sediment organic carbon and interstitial water is at or near equilibrium; (2) the concentration in either phase can be predicted using appropriate partition coefficients and the measured concentration in the other phase (assuming the freely-dissolved interstitial water concentration can be accurately measured); (3) organisms receive equivalent exposure from water-only exposures or from any equilibrated phase: either from interstitial water via respiration, from sediment via ingestion or other sediment-integument exchange, or from a mixture of exposure routes; (4) for nonionic chemicals, effect concentrations in sediments on an organic carbon basis can be predicted using the organic carbon partition coefficient (K_{OC}) and effects concentrations in water; (5) the FCV concentration is an appropriate effects concentration for freely-dissolved chemical in interstitial water; and (6) ESBs derived as the product of the K_{OC} and FCV are protective of benthic organisms. ESB concentrations presented in this document are expressed as μg chemical/g sediment organic carbon ($\mu g/g_{OC}$) and not on an interstitial water basis because (1) interstitial water is difficult to sample and (2) significant amounts of the dissolved chemical may be associated with dissolved organic carbon; thus, total concentrations in interstitial water may overestimate exposure.

Sediment benchmarks generated using the EqP approach are suitable for use in providing technical information to regulatory agencies because they are:

1. Numeric values
2. Chemical specific
3. Applicable to most sediments
4. Predictive of biological effects
5. Protective of benthic organisms

ESBs are derived using the available scientific data to assess the likelihood of significant environmental effects to benthic organisms from chemicals in sediments in the same way that the WQC are derived using the available scientific data to assess the likelihood of significant environmental effects to organisms in the water column. As such, ESBs are intended to protect benthic organisms from the effects of chemicals associated with sediments and, therefore, only apply to sediments permanently inundated with water, to intertidal sediment, and to sediments inundated periodically for durations sufficient to permit development of benthic assemblages. ESBs should not be applied to occasionally inundated soils containing terrestrial organisms, nor should they be used to address the question of possible contamination of upper trophic level organisms or the synergistic, additive, or antagonistic effects of multiple chemicals. The application of ESBs under these conditions may result in values lower or higher than those presented in this document.

ESB values presented herein are the concentrations of endrin in sediment that will not adversely affect most benthic organisms. It is recognized that these ESB values may need to be adjusted to account for future data. They may also need to be adjusted because of site-specific considerations. For example, in spill situations, where chemical equilibrium between water and sediments has not yet been reached, sediment chemical concentrations less than an ESB may pose risks to benthic organisms. This is because for spills, disequilibrium concentrations in interstitial and overlying water may be proportionally higher relative to sediment concentrations. Research has shown that the source or "quality" of total organic carbon (TOC) in the sediment does not affect chemical binding (DeWitt et al., 1992). However, the physical form of the chemical in the sediment may have an effect. At some sites, concentrations in excess of an ESB may not pose risks to benthic organisms because the compound may be a component of a particulate such as coal or soot, or exceed solubility such as undissolved oil or chemical. In these situations, an ESB would be overly protective of benthic organisms and should not be used unless modified using the procedures outlined in "Procedures for the Derivation of Site-Specific Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms" (U.S. EPA, 2003b). If the organic carbon has a low capacity (e.g., hair, sawdust, hide), an ESB would be underprotective. An ESB may also be underprotective where the toxicity of other chemicals are additive with an ESB chemical or where species of unusual sensitivity occur at the site.

This document presents the theoretical basis and the supporting data relevant to the derivation of ESBs for endrin. The data that support the EqP approach for deriving ESBs for nonionic organic chemicals are reviewed by Di Toro et al. (1991) and EPA (U.S. EPA, 2003a). Before proceeding through the following text, tables, and calculations, the reader should consider reviewing Stephan et al. (1985), U.S. EPA (1985) and U.S. EPA (2003a).

1.2 General Information: Endrin

Endrin is the common name of a "broad spectrum" organochlorine insecticide/rodenticide. It was formulated for use as an emulsifiable concentrate, as a wettable or dustable powder, or as a granular product. It has been used with a variety of crops including cotton, tobacco, sugar cane, rice, and ornamentals. One of its major uses in the United States was for control of Lepidoptera larvae on cotton. During the 1970's and early 1980's its use was increasingly restricted until it was banned on October 10, 1984, in part as a result of its observed toxicity to non-target organisms, bioaccumulation potential, and persistence [49 CFR 42792 (October 24, 1984)].

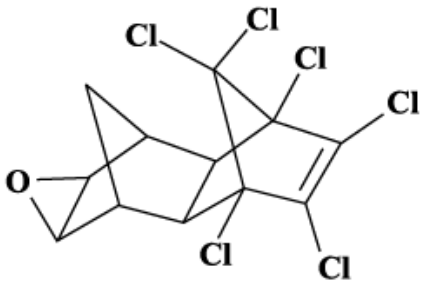
Structurally, endrin is a cyclic hydrocarbon having a chlorine substituted methanobridge structure (Figure 1-1). It is similar to dieldrin, an endo-endo stereoisomer, and has similar physicochemical properties, except that it is more easily degraded in the environment (Wang, 1988). Endrin is a colorless crystalline solid at room temperature, having a melting point of about 235°C and specific gravity of 1.7 g/cc at 20°C. It has a vapor pressure of 0.026 mPa (25°C) (Hartley and Kidd, 1987).

Endrin is toxic to non-target aquatic organisms, birds, bees, and mammals (Hartley and Kidd, 1987). The acute toxicity of endrin ranges from genus mean acute values (GMAVs) of 0.15 to 716.88 μg/L for freshwater organisms and 0.037 to 790 μg/L for saltwater organisms (Appendix A). There is little difference between the acute and chronic toxicity of

endrin to aquatic species; acute–chronic ratios (ACRs) range from 1.881 to 4.720 for three species (see Table 3-2 in Section 3.3). Endrin bioconcentrates in aquatic animals from 1,450 to 10,000 times the concentration in water (U.S. EPA, 1980). The WQC for endrin (U.S. EPA, 1980) was derived using a Final Residue Value (FRV) calculated using bioconcentration data and the Food and Drug Administration (FDA) action level to protect marketability of fish and shellfish; therefore, the WQC is not "effects based." In contrast, the ESB for endrin is effects based. It is calculated from the FCV derived in Section 3.

1.3 Applications of Sediment Benchmarks

ESBs are meant to be used with direct toxicity testing of sediments as a method of evaluation assuming the toxicity testing species is sensitive to the chemical of interest. They provide a chemical-by-chemical specification of what sediment concentrations are protective of benthic aquatic life. The EqP method should be applicable to nonionic organic chemicals with a K_{OW} above 3.0. Examples of other chemicals to which this methodology applies include dieldrin, metal mixtures (Cd, Cu, Pb, Ni, Ag, Zn), and polycyclic aromatic hydrocarbon (PAH) mixtures.



MOLECULAR FORMULA	**$C_{12}H_8Cl_6O$**
MOLECULAR WEIGHT	**380.93**
DENSITY	**1.70 g/cc (20°C)**
MELTING POINT	**235°C**
PHYSICAL FORM	**Colorless crystal**
VAPOR PRESSURE	**0.026 mPa (25°C)**

CAS NUMBER:	**72-20-8**
TSL NUMBER:	**IO 15750**
COMMON NAME:	**Endrin (also endrine and nendrin)**
TRADE NAME:	**Endrex (Shell); Hexadrin**
CHEMICAL NAME:	**1,2,3,4,10,10, hexachloro-1R, 4S, 4aS, 5nS, 6, 7R, 8R, 8aR-octahydro-6, 7-epoxy-1, 4:5, 8-dimethanonaphthalene (IUPAC) or Hexachloroepoxy-octahydro-endo-endo-di methanonaphthalene**

Figure 1-1. Chemical structure and physical-chemical properties of endrin (from Hartley and Kidd, 1987).

For the toxic chemicals addressed by the ESB documents Tier 1 (U.S. EPA, 2003c, d, e, and this document) and Tier 2 (U.S. EPA, 2003f) values were developed to reflect the differing degrees of data availability and uncertainty. Tier 1 ESBs are more scientifically rigorous and data intensive than Tier 2 ESBs. The minimum requirements to derive a Tier 1 ESB include: (1) Each chemical's organic carbon-water partition coefficient (K_{OC}) is derived from the octanol-water partition coefficient (K_{OW}) obtained using the SPARC (SPARC Performs Automated Reasoning in Chemistry) model (Karickhoff et al., 1991) and the K_{OW}-K_{OC} relationship from DiToro et al. (1991). This K_{OC} has been demonstrated to predict the toxic sediment concentration from the toxic water concentration with less uncertainty than K_{OC} values derived using other methods. (2) The FCV is updated using the most recent toxicological information and is based on the National WQC Guidelines (Stephan et al., 1985). (3) EqP confirmation tests are conducted to demonstrate the accuracy of the EqP prediction that the K_{OC} multiplied by the effect concentration from a water-only toxicity test predicts the effect concentration from sediment tests (Swartz, 1991; DeWitt et al., 1992). Using these specifications, Tier 1 ESBs have been derived for the insecticide endrin in this document, the nonionic organic insecticide dieldrin (U.S. EPA, 2003c), metals mixtures (U.S. EPA, 2003d), and polycyclic aromatic hydrocarbon (PAH) mixtures (U.S. EPA, 2003e). In comparison, the minimum requirements for a Tier 2 ESB (U.S. EPA, 2003f) are less rigorous: (1) The K_{OW} for the chemical that is used to derive the K_{OC} can be from slow-stir, generator column, shake flask, SPARC or other sources. (2) FCVs can be from published or draft WQC documents, the Great Lakes Initiative or developed from AQUIRE. Secondary chronic values (SCV) from Suter and Mabrey (1994) or other effects concentrations from water-only tests can be also used. (3) EqP confirmation tests are recommended, but are not required for the development of Tier 2 ESBs. Because of these lesser requirements, there is greater uncertainty in the EqP prediction of the sediment effect concentration from the water-only effect concentration, and in the level of protection afforded by Tier 2 ESBs. Examples of Tier 2 ESBs for nonionic organic chemicals are found in U.S. EPA (2003f).

1.4 Overview

Section 1 provides a brief review of the EqP methodology and a summary of the physical-chemical properties and aquatic toxicity of endrin. Section 2 reviews a variety of methods and data useful in deriving partition coefficients for endrin and includes the K_{OC} recommended for use in deriving endrin ESB_{WQC}s. Section 3 reviews aquatic toxicity data contained in the endrin WQC document (U.S. EPA, 1980) and new data that were used to calculate the FCV used in this document to derive ESB_{WQC} concentrations. In addition, the comparative sensitivity of benthic and water column species is examined, and justification is provided for use of the FCV for endrin in the derivation of ESB_{WQC}s. Section 4 reviews data on the toxicity of endrin in sediments, the need for organic carbon normalization of endrin sediment concentrations, and the accuracy of the EqP prediction of sediment toxicity using K_{OC} and an effect concentration in water. Data from Sections 2, 3, and 4 were used in Section 5 as the basis for the derivation of the ESB_{WQC}s for endrin and its uncertainty. ESB_{WQC}s for endrin are then compared with two databases on endrin's environmental occurrence in sediments. Section 6 concludes with the sediment benchmarks for endrin and their application and interpretation. The references cited in this document are listed in Section 7.

Section 2

Partitioning

2.1 Description of EqP Methodology

ESBs are the numeric concentrations of individual chemicals that are intended to be predictive of biological effects, protective of the presence of benthic organisms, and applicable to the range of natural sediments from lakes, streams, estuaries, and near-coastal marine waters. For nonionic organic chemicals, ESBs are expressed as μg chemical/g_{OC} and apply to sediments having $\geq$0.2% organic carbon by dry weight. A brief overview follows of the concepts that underlie the EqP methodology for deriving ESBs. The methodology is discussed in detail in "Technical Basis for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: Nonionic Organics" (U.S. EPA, 2003a), hereafter referred to as the ESB Technical Basis Document.

Bioavailability of a chemical at a particular sediment concentration often differs from one sediment type to another. Therefore, a method is necessary to determine ESBs based on the bioavailable chemical fraction in a sediment. For nonionic organic chemicals, the concentration–response relationship for the biological effect of concern can most often be correlated with the interstitial water (i.e., pore water) concentration (μg chemical/L interstitial water) and not with the sediment chemical concentration (μg chemical/g sediment) (Di Toro et al., 1991). From a purely practical point of view, this correlation suggests that if it were possible to measure the interstitial water chemical concentration, or predict it from the total sediment concentration and the relevant sediment properties, then that concentration could be used to quantify the exposure concentration for an organism. Thus, knowledge of the partitioning of chemicals between the solid and liquid phases in a sediment is a necessary component for establishing ESBs. For this reason, the methodology described below is called the EqP method. As stated above, an ESB can be derived using any given level of protection, in the following example the FCV from the endrin WQC is used.

The ESB Technical Basis Document shows that benthic species, as a group, have sensitivities similar to all benthic and water column species tested (taken as a group) to derive the WQC concentration for a wide range of chemicals. The data showing this for endrin are presented in Section 3.4. Thus, an ESB can be established using the FCV, calculated based on the WQC Guidelines (Stephan et al., 1985), as the acceptable effect concentration in interstitial or overlying water (see Section 5). The partition coefficient can then be used to relate the interstitial water concentration (i.e., the calculated FCV) to the sediment concentration via the partitioning equation. This acceptable concentration in sediment is an ESB_{WQC}.

An ESB is calculated as follows. Let FCV (μg/L) be the acceptable concentration in water for the chemical of interest, then compute an ESB using the partition coefficient, K_P (L/$kg_{sediment}$), between sediment and water

$$ESB_{WQC} = K_P \, FCV \qquad (2\text{-}1)$$

This is the fundamental equation used to generate an ESB_{WQC}. Its utility depends on the existence of a methodology for quantifying K_P.

Organic carbon appears to be the dominant sorption phase for nonionic organic chemicals in naturally occurring sediments and, thus, controls the bioavailability of these compounds in sediments. Evidence for this can be found in numerous toxicity tests, bioaccumulation studies, and chemical analyses of interstitial water and sediments (Di Toro et al., 1991). The evidence for endrin is discussed in this section and in Section 4. The organic carbon binding of a chemical in sediment is a function of that chemical's K_{OC} and the weight fraction of organic carbon (f_{OC}) in the sediment. The relationship is as follows

$$K_P = f_{OC} \, K_{OC} \qquad (2\text{-}2)$$

It follows that

$$ESB_{WQCOC} = K_{OC} \, FCV \qquad (2\text{-}3)$$

where ESB_{WQCOC} is an ESB_{WQC} on a sediment organic carbon basis. For nonionic organics, "ESB_{WQC}" usually refers to a value that is organic carbon–normalized (more formally ESB_{WQCOC}) unless otherwise specified.

K_{OC} is not usually measured directly (although it can be done; see Section 2.3). Fortunately, K_{OC} is closely related to the octanol-water partition coefficient (K_{OW}) (Equation 2-5), which has been measured for many compounds and can be measured very accurately. The next section reviews the available information on the K_{OW} for endrin.

2.2 Determination of K_{OW} for Endrin

Several approaches have been used to determine K_{OW} for the derivation of an ESB, as discussed in the ESB Technical Basis Document. In an examination of the literature, primary references were found listing measured $\log_{10}K_{OW}$ values for endrin ranging from 4.40 to 5.19 and estimated $\log_{10}K_{OW}$ values ranging from 3.54 to 5.60 (Table 2-1). Karickhoff and Long (1995, 1996) established a protocol for recommending K_{OW} values for uncharged organic chemicals based on the best available measured, calculated, and estimated data. The recommended $\log_{10}K_{OW}$ value of 5.06 for endrin from Karickhoff and Long (1995) is used to derive ESBs for endrin.

2.3 Derivation of K_{OC} from Adsorption Studies

Two types of experimental measurements of K_{OC} are available. The first type involves experiments designed to measure the partition coefficient in particle suspensions. The second type is from sediment toxicity tests in which sediment endrin, sediment organic carbon (OC) and freely-dissolved endrin in interstitial water were used to compute K_{OC}; endrin associated with dissolved organic carbon (DOC) was not included.

2.3.1 K_{OC} *from Particle Suspension Studies*

Laboratory studies to characterize sorption are generally conducted using particle suspensions. The high concentrations of solids and turbulent conditions necessary to keep the mixture in suspension make data interpretation difficult as a result of the particle interaction effect. This effect suppresses the partition coefficient relative to that observed for undisturbed sediments (Di Toro, 1985; Mackay and Powers, 1987).

Based on analysis of an extensive body of experimental data for a wide range of compound types and experimental conditions, the particle interaction model (Di Toro, 1985) yields the following relationship for estimating K_p

$$K_P = \frac{f_{OC}\, K_{OC}}{1 + m f_{OC}\, K_{OC}/\upsilon_x} \qquad (2\text{-}4)$$

where m is the particle concentration in the suspension (kg/L) and υ_X, an empirical constant, is 1.4. The K_{OC} is given by

$$\log_{10}K_{OC} = 0.00028 + 0.983 \log_{10}K_{OW} \qquad (2\text{-}5)$$

Figure 2-1 compares observed partition coefficient data for the reversible component with predicted values estimated with the particle interaction model (Equations 2-4 and 2-5) for a wide range of compounds

Table 2-1. Endrin measured and estimated $\log_{10}K_{OW}$ values

Method	$Log_{10}K_{OW}$	Reference
Measured	4.40	Rapaport and Eisenreich, 1984
Measured	4.92	Ellington and Stancil, 1988
Measured	5.01	Eadsforth, 1986
Measured	5.19	De Bruijn et al., 1989
Estimated	3.54	Mabey et al., 1982
Estimated	5.40	Karickhoff et al., 1989
Estimated	5.60	Neeley et al., 1974

(Di Toro, 1985). The observed partition coefficient for endrin using adsorption data (Sharom et al., 1980) is highlighted on this plot. The observed $\log_{10}K_p$ of 2.04 reflects significant particle interaction effects. The observed partition coefficient is about nine times lower than the value expected in the absence of particle effects (i.e., $\log_{10}K_p = 2.98$ from $f_{OC}K_{OC} = 958$ L/kg). In the absence of particle effects, K_{OC} is related to K_{OW} via Equation 2-5. For $\log_{10}K_{OW} = 5.06$ (see Section 2.2), this expression results in an estimate of $\log_{10}K_{OC} = 4.97$.

2.3.2 K_{OC} *from Sediment Toxicity Tests*

Measurements of K_{OC} were available from the sediment toxicity tests using endrin (Nebeker et al., 1989; Schuytema et al., 1989; Stehly, 1992). These tests used different freshwater sediments having a range of organic carbon contents of 0.07% to 11.2% (see Table 4-1; Appendix B). Endrin concentrations were measured in the sediment and interstitial waters, providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. In the case of the data reported by Schuytema et al. (1989), the concentration of endrin in the overlying water at the end of the 10-day experiment was used. Nebeker et al. (1989) demonstrated in their experiments, which were static and run in the same way as those of Schuytema et al. (1989), that overlying water and interstitial water endrin concentrations were similar. Figure 2-2A is a plot of the organic carbon–normalized sorption isotherm for endrin, where the sediment endrin concentration ($\mu g/g_{OC}$) is plotted versus interstitial water concentration ($\mu g/L$). The data used to make this plot are included in Appendix B. The line of unity slope corresponding to the $\log_{10}K_{OC} = 4.97$ derived from the endrin $\log_{10}K_{OW}$ of 5.06 from Karickhoff and Long (1995) is compared with the data. A probability plot of the observed experimental $\log_{10}K_{OC}$ values is shown in Figure 2-2B. The $\log_{10}K_{OC}$ values were approximately normally distributed, with a mean of $\log_{10}K_{OC} = 4.67$ and a standard error of the mean (SE) of 0.04. This value agrees with the $\log_{10}K_{OC} = 4.97$, which was computed using the endrin $\log_{10}K_{OW}$ of 5.06 from Karickhoff and Long (1995) using Equation 2-5.

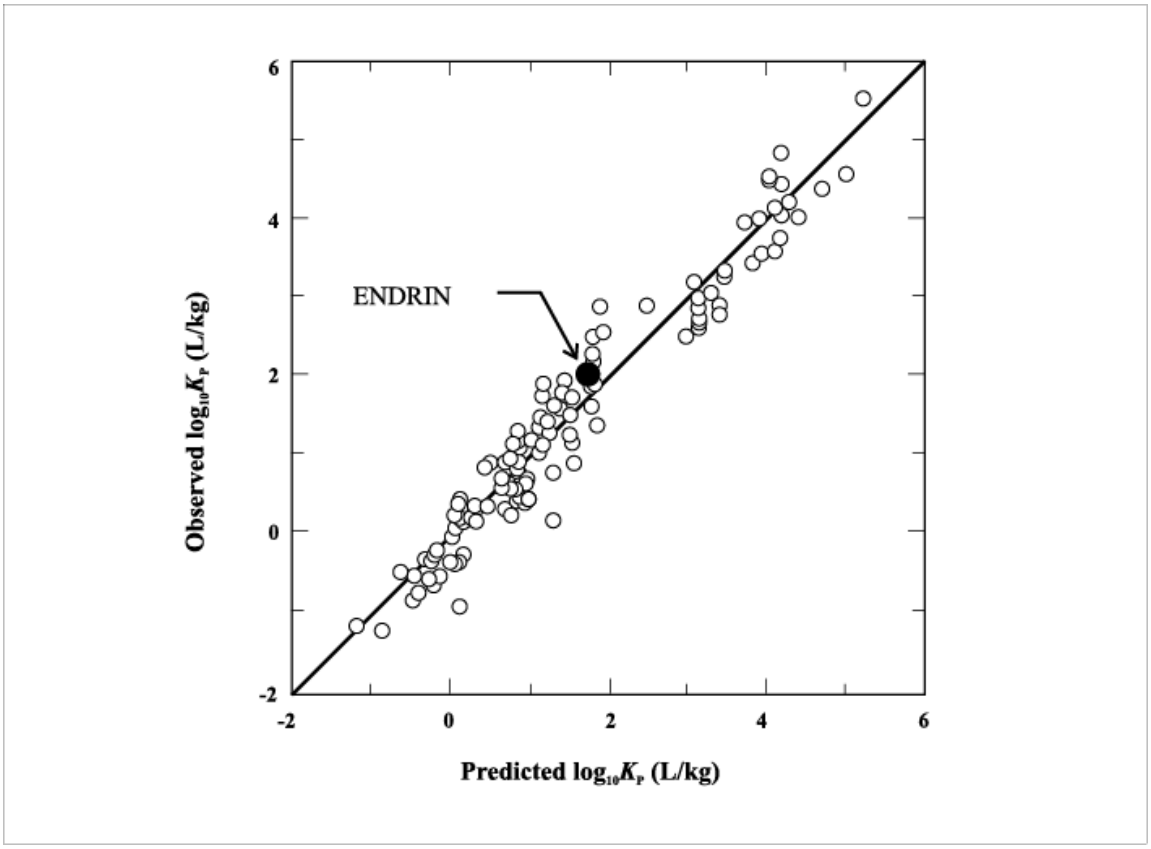


Figure 2-1. Observed versus predicted partition coefficients for nonionic organic chemicals using Equation 2-4 (figure from DiToro, 1985). Endrin datum is highlighted (Sharom et al., 1980).

2.4 Summary of Derivation of K_{OC} for Endrin

The K_{OC} selected to calculate ESBs for endrin were based on the regression of $\log_{10}K_{OC}$ to $\log_{10}K_{OW}$ (Equation 2-5) using the endrin $\log_{10}K_{OW}$ of 5.06 from Karickhoff and Long (1995). This approach, rather than use of the K_{OC} from the toxicity tests, was adopted because the regression equation is based on the most robust dataset available that spans a broad range of chemicals and particle types, thus encompassing a wide range of K_{OW} and f_{OC} values. The regression equation yielded a $\log_{10}K_{OC}$ of 4.97. This value is comparable to the $\log_{10}K_{OC}$ of 4.67 measured in the sediment toxicity tests.

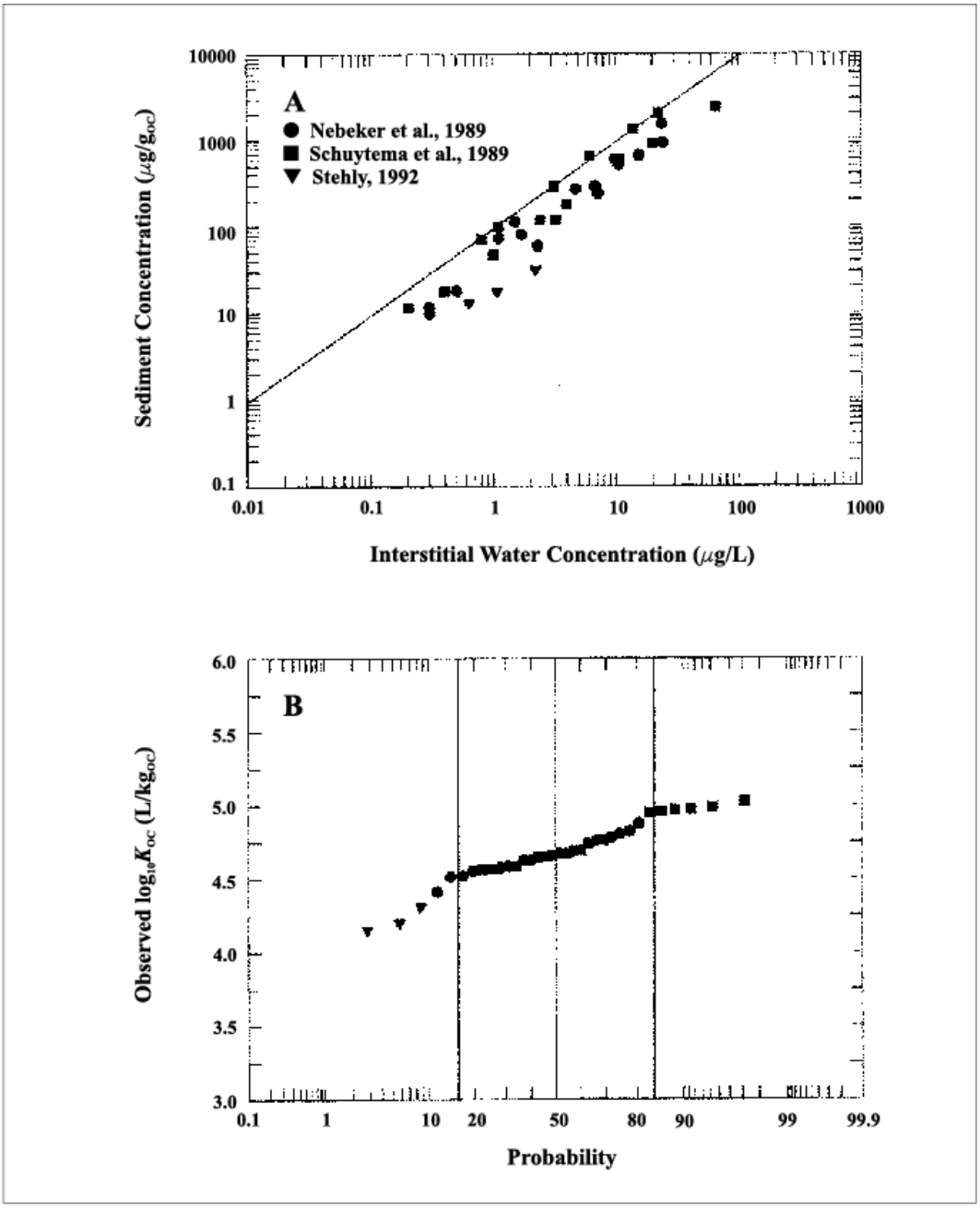


Figure 2-2. Organic carbon–normalized sorption isotherm for endrin (A) and probability plot of K_{OC} (B) from sediment toxicity tests (Nebeker et al., 1989; Schuytema et al., 1989; Stehly, 1992). The solid line represents the relationship predicted with a $\log_{10}K_{OC}$ of 4.97.

Section 3

Toxicity of Endrin in Water Exposures

3.1 Derivation of Endrin WQC

The example used in Section 2 for the EqP method for derivation of ESB_{WQC}s for endrin uses the WQC FCV and K_{OC} to estimate the maximum concentration of nonionic organic chemical in sediments, expressed on an organic carbon basis, that will not cause adverse effects to benthic organisms. For this document, life-stages of species classified as benthic are either species that live in the sediment (infaunal) or on the sediment surface (epibenthic) and obtain their food from either the sediment or water column (U.S. EPA, 2003a). In this section, the FCV from the endrin WQC document (U.S. EPA, 1980) is revised using new aquatic toxicity test data, and the use of this FCV is justified as the appropriate effects concentration for the derivation of endrin ESB_{WQC}s.

3.2 Acute Toxicity in Water Exposures

A total of 104 standard acute toxicity tests with endrin have been conducted on 42 freshwater species from 34 genera (Figure 3-1; Appendix A). Overall GMAVs ranged from 0.15 to 180 μg/L. Fishes, stoneflies, caddisflies, dipterans, mayflies, glass shrimp, isopods, ostracods, amphipods, and damselflies were most sensitive; overall GMAVs for the most sensitive genera of these taxa range from 0.15 to 4.6 μg/L. This database contains 39 tests on the benthic life-stages of 25 species from 22 genera (Figure 3-1; Appendix A). Benthic organisms were both among the most sensitive and the most resistant freshwater species to endrin. Of the epibenthic species, stoneflies, caddisflies, fish, mayflies, glass shrimp, damselflies, amphipods, and dipterans were most sensitive; GMAVs ranged from $>$0.18 to 12 μg/L. Infaunal species tested included stoneflies, mayflies, dipterans, a midge, an oligochaete worm, and an ostracod; GMAVs ranged from 0.83 μg/L for the midge, *Tanytarsus,* to $>$165 μg/L for the oligochaete, *Lumbriculus*.

A total of 37 acute toxicity tests were conducted on 21 saltwater species from 19 genera (Figure 3-2; Appendix A). Overall GMAVs ranged from 0.037 to 790 μg/L. Fishes and a penaeid shrimp were most sensitive; however, only 7 of the 21 species tested were invertebrates. Results from 25 tests on benthic life-stages of 13 species from 11 genera are contained in this database (Figure 3-2; Appendix A). Benthic organisms were among both the most sensitive and most resistant saltwater genera to endrin. The most sensitive benthic species was the commercially important pink shrimp, *Penaeus duorarum*, with a measured flow-through 96-hour LC50 of 0.037 μg/L. The LC50 represents the chemical concentrations estimated to be lethal to 50% of the test organisms within a specified time period. Other benthic species for which there are data appeared less sensitive, with GMAVs ranging from 0.094 to 12 μg/L.

3.3 Chronic Toxicity in Water Exposures

Life-cycle toxicity tests have been conducted with the freshwater flagfish (*Jordanella floridae*) and fathead minnow (*Pimephales promelas*) and with the saltwater sheepshead minnow (*Cyprinodon variegatus*) and grass shrimp (*Palaemonetes pugio*). Each of these species, except for *P. promelas*, has one or more benthic life-stages.

Two life-cycle toxicity tests have been conducted with *J. floridae* (Table 3-1). The concentration–response relationships were almost identical among the tests. Hermanutz (1978) observed an 8% reduction in growth (length) and a 79% reduction in number of eggs spawned per female in 0.30 μg/L endrin relative to response of control fish; progeny were unaffected (Table 3-1). Neither parental nor progeny (F_1) generation *J. floridae* were significantly affected when exposed to endrin concentrations from 0.051 to 0.22 μg/L. The chronic value from this test was 0.2569. Combined with the 96-hour companion acute value of 0.85 μg/L (Hermanutz et al., 1985), the acute-chronic ratio (ACR) for this test is 3.309 (Table 3-2).

In the second life-cycle test, Hermanutz et al. (1985) observed a 51% decrease in reproduction in parental fish exposed to 0.29 μg/L endrin, and

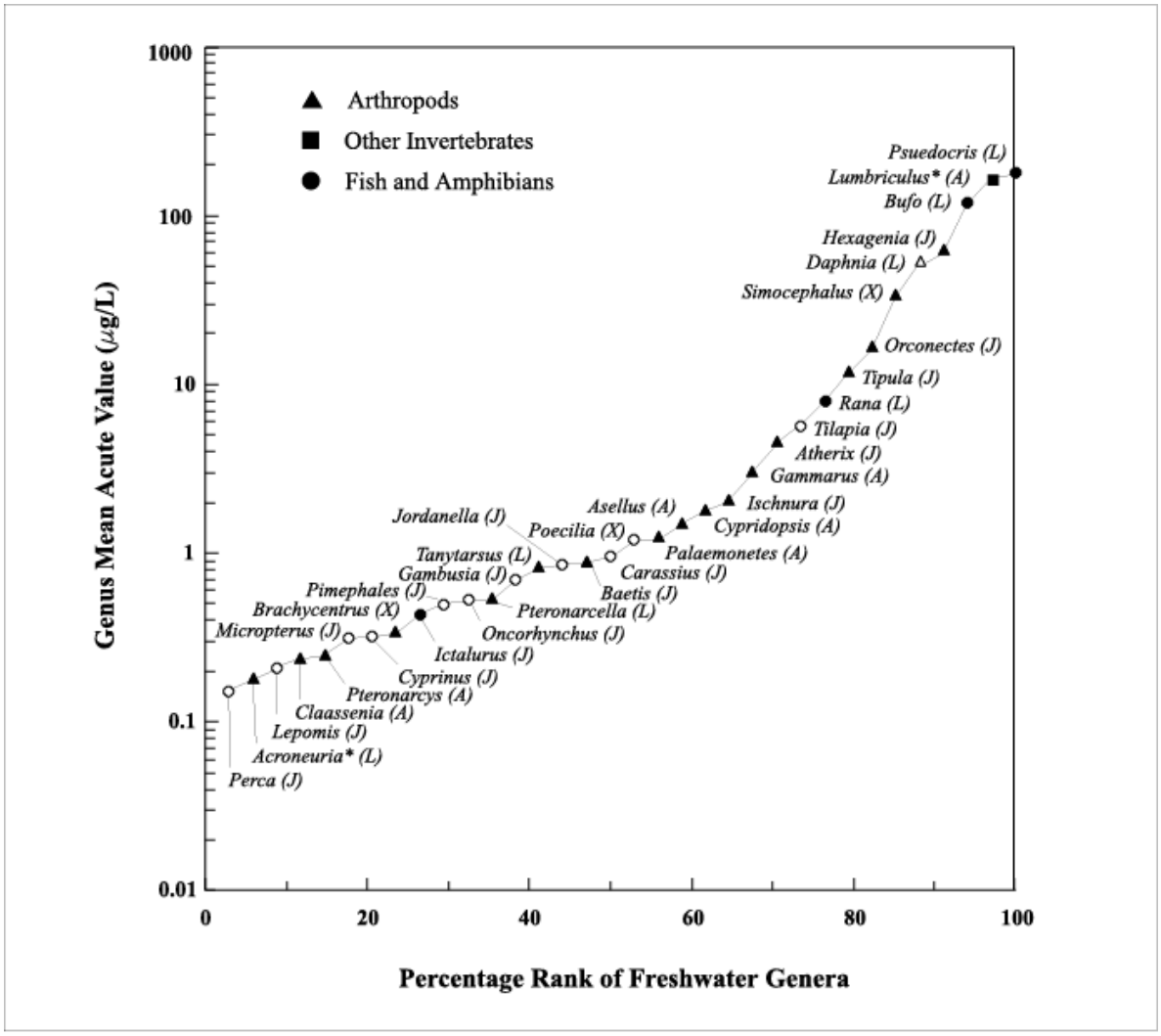


Figure 3-1. Genus mean acute values from water-only acute toxicity tests using freshwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. Asterisks indicate greater than values. A = adult, J = juvenile, L = larvae, X = unspecified life-stage.

reductions of 73% in survival, 18% in (growth) length, and 92% in numbers of eggs per female in 0.39 μg/L. No significant effects were detected in parental or progeny generation flagfish in 0.21 μg/L. The chronic value from this test was 0.2468. Combined with the 96-hour companion acute value of 0.85 μg/L (Hermanutz et al., 1985), the ACR for this test is 3.444. The geometric mean of these two ACRs is 3.376.

The effect of endrin on *P. promelas* in a life-cycle test was only marginally enhanced when exposure was via water and diet versus water-only exposures (Jarvinen and Tyo, 1978). Parental fish in 0.25 μg/L in water-only exposures exhibited about 60% mortality relative to controls. Mortality of F_1 progeny was 70% in 0.14 μg/L, the lowest concentration tested, and 85% in 0.25 μg/L. Tissue concentrations increased marginally in fish exposed to the water and diet treatment relative to fish in water-only exposures. Effects were observed at all concentrations tested, so the chronic value for this test is considered to be <0.14 μg/L. No ACR from this test can be calculated because no acute value from matching dilution water is available.

One saltwater invertebrate species, *P. pugio*, has been exposed to endrin in a partial life-cycle toxicity test (Tyler-Schroeder, 1979). Mortality of parental generation shrimp generally increased as endrin concentrations increased from 0.11 to 0.79 μg/L. Onset

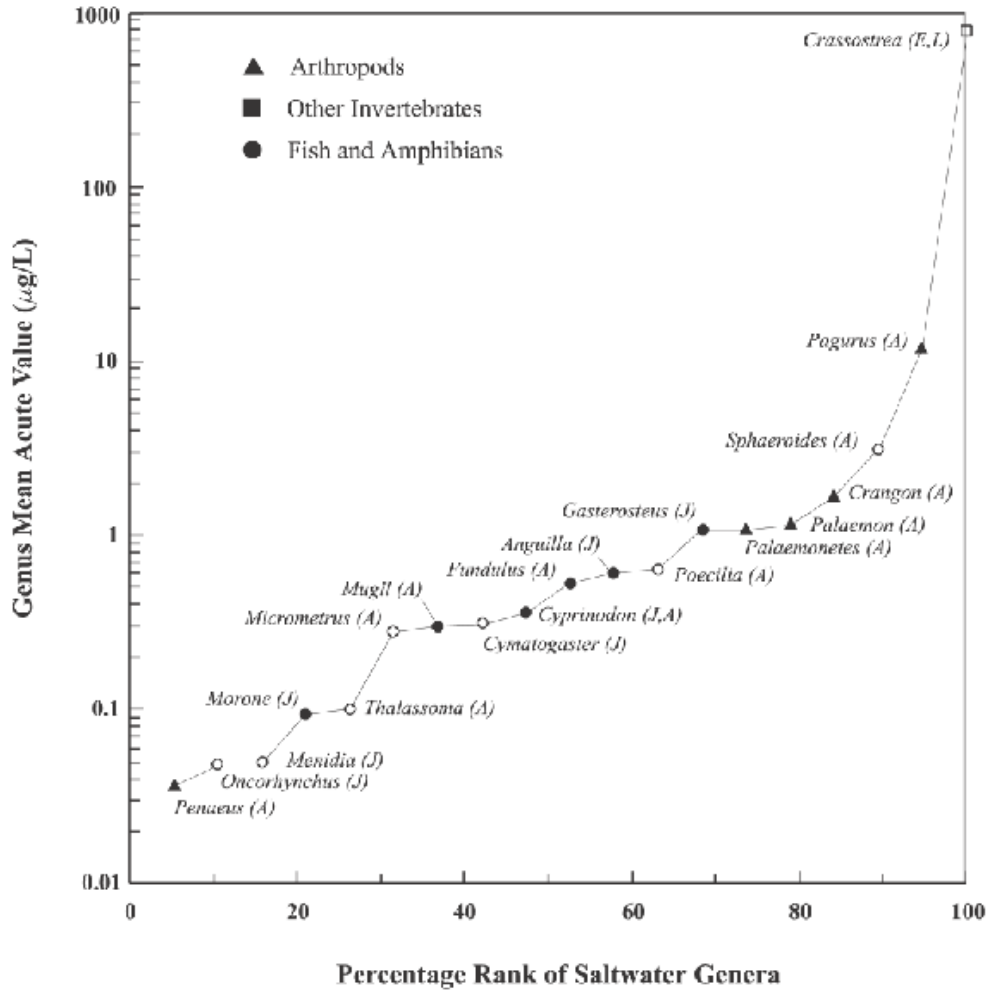


Figure 3-2. Genus mean acute values from water-only acute toxicity tests using saltwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. A = adult, E = embryo, J = juvenile, L = larvae.

of spawning was delayed, duration of spawning was lengthened, and the number of female *P. pugio* spawning was less in all exposure concentrations from 0.03 to 0.79 μg/L. These effects on reproduction may not be important because embryo production and hatching success were apparently not affected. Larval mortality and time to metamorphosis increased and growth of juvenile progeny decreased in endrin concentrations $\geq$0.11 μg/L. The chronic value from this test was 0.07416. Combined with the 96-hour companion acute value of 0.35 μg/L (Tyler-Schroeder, 1979), the ACR for this test is 4.720.

C. variegatus exposed to endrin in a life-cycle toxicity test (Hansen et al., 1977) were affected at endrin concentrations similar to those affecting the two freshwater fishes described above. Embryos exposed to 0.31 and 0.72 μg/L endrin hatched early, and all fry exposed to 0.72 μg/L and about half of those exposed to 0.31 μg/L died. Females died during spawning, fewer eggs were fertile, and survival of exposed progeny decreased in 0.31 μg/L. No significant effects were observed on survival, growth, or reproduction in fish exposed to 0.027 to 0.12 μg/L endrin. The chronic value from this test was 0.1929. Combined with the 96-hour companion acute value of 0.3629 μg/L (Hansen et al., 1977; Schimmel et al., 1975), the ACR for this test is 1.881.

The difference between acute and chronic toxicity of endrin was small (Table 3-2). ACR values were 3.309 and 3.444 for *J. floridae*, 4.720 for *P. pugio,* and 1.881 for *C. variegatus*. The final ACR (FACR) was 3.106 for both freshwater and saltwater species. Long-term exposures, not classed as "chronic" in the National WQC Guidelines (Stephan et al., 1985), also

Table 3-1. Test-specific data for chronic sensitivity of freshwater and saltwater organisms to endrin

Common Name, *Scientific Name*	Test[a]	Habitat[b] (life-stage)	Duration (days)	NOECs[c] (μg/L)	OECs[c] (μg/L)	Observed Effects (relative to controls)	Chronic Value (μg/L)	Reference
Freshwater Species								
Flagfish, *Jordanella floridae*	LC	E (E,L) W (J,A)	110	0.051-0.22	0.30	8% reduction in growth, 79% reduction in reproduction	0.2569	Hermanutz, 1978
Flagfish, *Jordanella floridae*	LC	E (E,L) W (J,A)	140	0.21	0.29, 0.39	51-92% reduction in reproduction, 73% decrease in survival, 18% reduction in growth	0.2468	Hermanutz et al., 1985
Fathead minnow, *Pimephales promelas*	LC	W (E,L,J,A)	300	<0.14	0.14-0.25	60% decrease in adult survival, 70-85% decrease in progeny survival	<0.14	Jarvinen and Tyo, 1978
Saltwater Species								
Grass shrimp, *Palaemonetes pugio*	PLC	W (L) E,W (E,J,A)	145	0.03, 0.05	0.11-0.79	38-100% decrease in adult survival, 26-94% reduction in progeny growth	0.07416	Tyler-Schroeder, 1979
Sheepshead minnow, *Cyprinodon variegatus*	LC	E (E) E,W (J,A)	175	0.027-0.12	0.31, 0.72	48-100% decrease in survival; 15% reduction in growth and in adult reproduction; 87% decrease in progeny survival	0.1929	Hansen et al., 1977

[a]Test: LC = life-cycle, PLC = partial life-cycle, ELS = early life-stage.
[b]Habitat: I = infauna, E = epibenthic, W = water column. Life-stage: E = embryo, L = larval, J = juvenile, A = adult.
[c]NOECs = no observed effect concentrations; OECs = observed effect concentrations.

indicated little difference between acute and chronic toxicity of endrin. These include tests with the caddisfly, *Brachycentrus americanus*; stonefly, *Pteronarcys dorsata* (Anderson and DeFoe, 1980); bluntnose minnow, *Pimephales notatus* (Mount, 1962); fathead minnow, *P. promelas* (Jarvinen et al., 1988); brown bullhead, *Ictalurus melas* (Anderson and DeFoe, 1980); largemouth bass, *Micropterus salmoides* (Fabacher, 1976); spot, *Leiostomus xanthurus* (Lowe, 1966); and mummichog, *Fundulus heteroclitus* (Eisler, 1970a).

The final acute value (FAV) derived from the overall GMAVs (Stephan et al., 1985) for freshwater organisms was 0.1803 μg/L. The FAV for saltwater species was 0.03282 μg/L (Table 3-2). The FCVs were used as the effect concentrations for calculating ESB_{WQC}s for protection of benthic species. The FCV of

Table 3-2. Summary of freshwater and saltwater acute and chronic values, acute-chronic ratios, and derivation of final acute values, final acute-chronic ratios, and final chronic values for endrin

Common Name, *Scientific Name*	Acute Value (μg/L)	Chronic Value (μg/L)	Acute-Chronic Ratio (ACR)	Species Mean Acute-Chronic Ratio (SMACR)
Freshwater Species				
Flagfish, *Jordanella floridae*	0.85	0.2569	3.309	—
Flagfish, *Jordanella floridae*	0.85	0.2468	3.444	3.376
Fathead minnow, *Pimephales promelas*		<0.14[a]		
Saltwater Species				
Grass shrimp, *Palaemonetes pugio*	0.35	0.07416	4.720	4.720
Sheepshead minnow, *Cyprinodon variegatus*	0.3629[b]	0.1929	1.881	1.881

[a]Not used in calculation of SMACR or FACR because acute value from matching dilution water is not available.
[b]Geometric mean of 96-hour LC50 values from three flow-through measured tests (0.34, 0.37, 0.38 μg/L) on fry or juvenile fish from Hansen et al. (1977) and Schimmel et al. (1975). These tests were performed in the same dilution water as the chronic test.

Freshwater:
Final acute value = 0.1803 μg/L
Final acute-chronic ratio = 3.106
Final chronic value = 0.05805 μg/L

Saltwater:
Final acute value = 0.03282 μg/L
Final acute-chronic ratio = 3.106
Final chronic value = 0.01057 μg/L

0.05805 μg/L for freshwater organisms is the quotient of the FAV of 0.1803 μg/L and the FACR of 3.106. Similarly, the FCV for saltwater organisms of 0.01057 μg/L is the quotient of the FAV of 0.03282 μg/L and the FACR of 3.106.

3.4 Applicability of the WQC as the Effects Concentration for Derivation of Endrin ESB_{WQC}s

Use of the FCV as the effects concentration for calculation of ESB_{WQC}s assumes that benthic (infaunal and epibenthic) species, as a group, have sensitivities similar to all species tested to derive the WQC concentration as a group. Di Toro et al. (1991) and the ESB Technical Basis Document (U.S. EPA, 2003a) present data supporting the reasonableness of this assumption, over all chemicals for which there were published or draft WQC documents. The conclusion of similar sensitivity was supported by comparisons between (1) acute values for the most sensitive benthic species and acute values for the most sensitive water column species for all chemicals, (2) acute values for all benthic species and acute values for all species in the WQC documents across all chemicals after standardizing the LC50 values, (3) FAVs calculated for benthic species alone and FAVs calculated for all species in the WQC documents, and (4) individual chemical comparisons of benthic species versus all species. Only in this last comparison were endrin-specific comparisons of the sensitivity of benthic and all (benthic and water column) species conducted. The following paragraphs examine the data on the similarity of sensitivity of benthic and all species for endrin used in this comparison.

For endrin, benthic species account for 22 out of 34 genera tested in freshwater and 11 of 19 genera tested in saltwater (Figures 3-1, 3-2). An initial test of the difference between the freshwater and saltwater FAVs for all species (water column and benthic) exposed to endrin was performed using the approximate randomization (AR) method (Noreen, 1989). The AR method tests the significance level of a test statistic compared with a distribution of statistics generated from many random subsamples. The test statistic in this case was the difference between the freshwater

FAV, computed from the freshwater (combined water column and benthic) species LC50 values, and the saltwater FAV, computed from the saltwater (combined water column and benthic) species LC50 values (Table 3-3). In the AR method, the freshwater LC50 values and the saltwater LC50 values (see Appendix A) were combined into one dataset. The dataset was shuffled, then separated back so that randomly generated "freshwater" and "saltwater" FAVs could be computed. The LC50 values were separated back such that the number of LC50 values used to calculate the sample FAVs was the same as the number used to calculate the original FAVs. These two FAVs were subtracted and the difference used as the sample statistic. This was done many times so that the sample statistics formed a distribution representative of the population of FAV differences (Figure 3-3A). The test statistic was compared with this distribution to determine its level of significance. The null hypothesis was that the LC50 values composing the saltwater and freshwater databases were not different. If this were true, the difference between the actual freshwater and saltwater FAVs should be common to the majority of randomly generated FAV differences. For endrin, the test statistic occurred at the 99th percentile of the generated FAV differences. Because the probability was greater than 95%, the hypothesis of no significant difference in sensitivity for freshwater and saltwater species was rejected (Table 3-3). Note that greater than (>) values for GMAVs (see Appendix A) were omitted from the AR analyses for both freshwater versus saltwater and benthic versus combined water column and benthic organisms. This resulted in two endrin freshwater benthic organisms being omitted.

Because freshwater and saltwater species did not show similar sensitivity, separate tests were conducted for freshwater and saltwater benthic species. For the species from each water type, a test of difference in sensitivity for benthic and all (benthic and water column species combined, hereafter referred to as "WQC") organisms, was performed using the AR method. For this purpose, each life-stage of each test organism was assigned a habitat (Appendix A) using the criteria described in U.S. EPA (2003a). The test statistic in this case was the difference between the WQC FAV, computed from the WQC LC50 values, and the benthic FAV, computed from the benthic organism LC50 values. This was slightly different from the previous test for saltwater and freshwater species in that saltwater and freshwater species represented two separate groups. In this test, the benthic organisms were a subset of the WQC organisms set. In the AR method for this test, the number of data points coinciding with the number of benthic organisms was selected from the WQC dataset and the "benthic" FAV was computed. The original WQC FAV and the "benthic" FAV were then used to compute the difference statistic. This was done many times, and the resulting distribution was representative of the population of FAV difference statistics. The test statistic was compared with this distribution to determine its level of significance. The probability distribution of the computed FAV differences is shown in Figures 3-3B and 3-3C. The test statistic for this analysis occurred at the 7th percentile for freshwater organisms and the 68th percentile for saltwater organisms, and the hypothesis of no difference in sensitivity was accepted (Table 3-3). This analysis suggests that the FCV for endrin based on data from all tested species was an appropriate effects concentration for benthic organisms.

Table 3-3. Results of approximate randomization (AR) test for the equality of the freshwater and saltwater FAV distributions for endrin and AR test for the equality of benthic and combined benthic and water column WQC FAV distributions

Comparison	Habitat or Water Type[a,b]	AR Statistic[c]	Probability[d]
Freshwater vs Saltwater	Fresh (32) Salt (19)	0.149	99
Freshwater: Benthic vs Water Column + Benthic (WQC)	Benthic (21) WQC (32)	0.042	7
Saltwater: Benthic vs Water Column + Benthic (WQC)	Benthic (11) WQC (19)	0.012	68

[a]Values in parentheses are the number of LC50 values used in the comparison.
[b]Note that in both the freshwater vs. saltwater and benthic vs. WQC comparisons, greater than (>) values in Appendix A were omitted. This resulted in two endrin freshwater benthic organisms being omitted from the AR analysis.
[c]AR statistic = FAV difference between original compared groups.
[d]Probability that the theoretical AR statistic ≤ the observed AR statistic, given that the samples came from the same population.

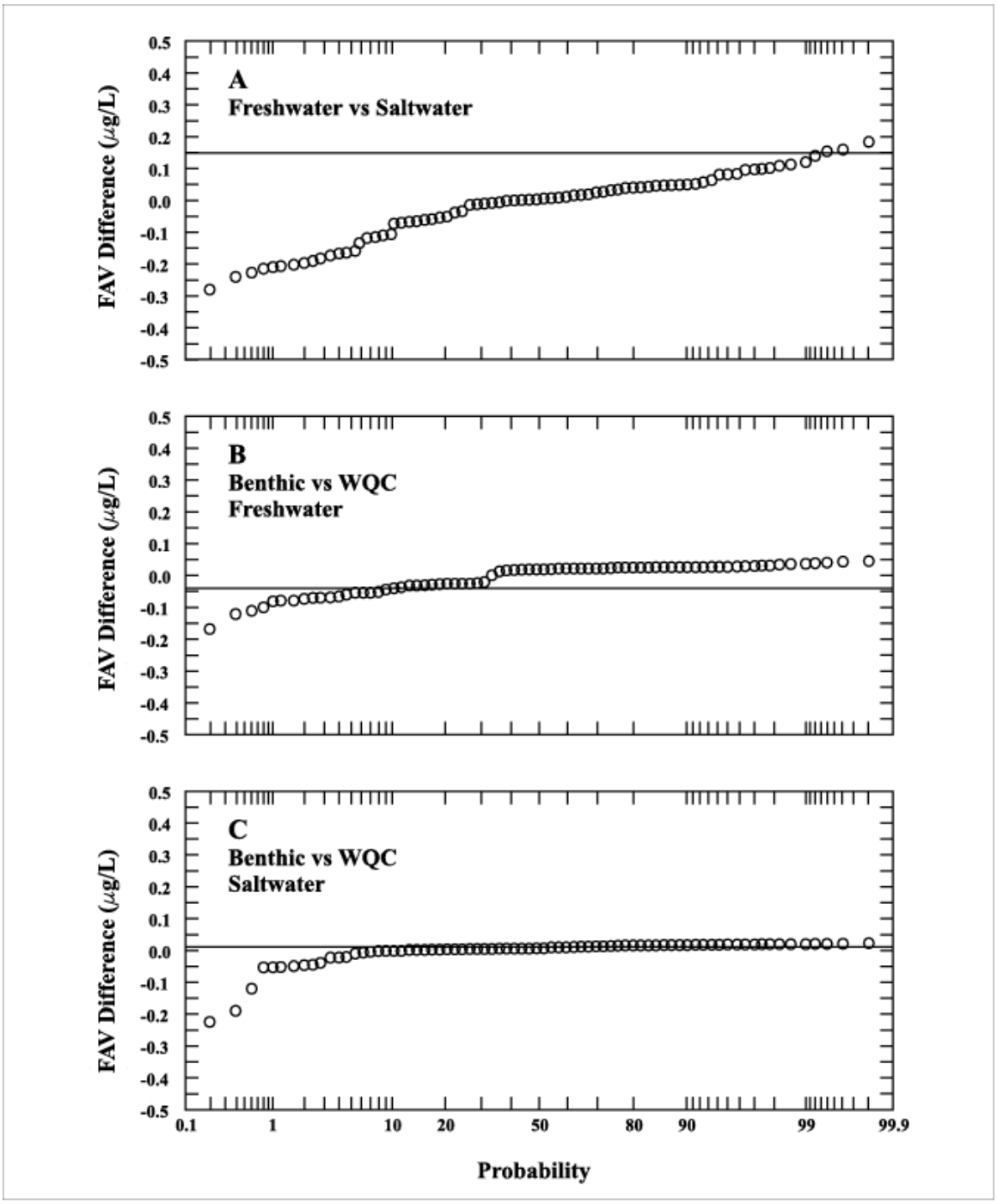


Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater versus saltwater (A), benthic versus WQC freshwater (B), and benthic versus WQC saltwater (C) data. The solid lines in the figure correspond to the FAV differences measured for endrin.

Section 4

Actual and Predicted Toxicity of Endrin in Sediment Exposures

4.1 Toxicity of Endrin in Sediments

The toxicity of endrin-spiked sediments was tested with four freshwater species (two oligochaetes—a lumbriculid worm and a tubificid worm—and two amphipods) and two saltwater species (a polychaete and the sand shrimp) (Table 4-1). The most common endpoint measured was mortality; however, impacts have been reported on sublethal endpoints such as growth, sediment avoidance, and sediment reworking rate. All concentrations of endrin in sediments or interstitial water where effects were observed were greater than ESB_{WQC} or FCV concentrations reported in this document. Details about exposure methodology are provided because sediment testing methodologies have not been standardized in the way that water-only toxicity test methodologies have. Generalizations across species or sediments are limited because of the limited number of experiments.

Keilty et al. (1988a,b) and Keilty and Stehly (1989) studied the effects on oligochaete worms of Lake Michigan sediments spiked with endrin. For all tests, sediments were dried, passed through a 0.25 mm sieve, reconstituted with lake water, spiked with endrin dissolved in acetone, and stirred for 24 hours. The water (containing the carrier) was aspirated off, new overlying water added, and sediments placed into individual beakers for 72 hours before the worms were added.

Keilty et al. (1988a) examined the effects of endrin-spiked sediment on sediment avoidance and mortality of two species of oligochaete worms in replicate 4-day exposures (Table 4-1). Four-day LC50 values for four tests with *Stylodrilus heringianus* averaged 2,110 μg endrin/g dry weight sediment and ranged from 1,050 to 5,400 μg endrin/g dry weight sediment. Four-day LC50 values for three tests with *Limnodrilus hoffmeisteri* averaged 3,390 μg/g dry weight sediment and ranged from 2,050 to 5,600 μg/g dry weight sediment. Four-day LC50 values from these tests averaged 194,000 $\mu g/g_{OC}$ for *L. hoffmeisteri* and 121,000 $\mu g/g_{OC}$ for *S. heringianus*. Data using this test method have demonstrated laboratory variabilities by a factor of 3 to 5 for the same sediment. Sediment avoidance was seen at much lower concentrations. Over all tests, burrowing was markedly reduced at $\geq$11.5 μg/g dry weight sediment and possibly at $\geq$0.54 μg/g dry weight sediment. EC50s, based on sediment avoidance, were 59.0 μg/g dry weight (3,371 $\mu g/g_{OC}$) for *L. hoffmeisteri* and 15.3 and 19.0 μg/g dry weight (874 and 1,086 $\mu g/g_{OC}$) sediment for two tests using *S. heringianus*. The EC50 represents the chemical concentration estimated to cause effects to 50% of the test organisms within a specified time period. Keilty et al. (1988b) observed 18% mortality of *S. heringianus* in 11.5 μg/g dry weight sediment after a 54-day exposure and 26% mortality in 42.0 μg/g dry weight sediment. The sediment reworking rate was reported to be significantly different from the control in sediments containing $\geq$0.54 μg/g dry weight sediment. Dry weights of worms in $\geq$2.33 μg/g dry weight sediment were reduced after 54 days. Keilty and Stehly (1989) observed no effect of a single, nominal concentration of 50 μg/g dry weight sediment on protein utilization by *S. heringianus* over a 69-day exposure period. However, dry weights of worms were significantly reduced.

Nebeker et al. (1989) and Schuytema et al. (1989) exposed the amphipod *Hyalella azteca* to two endrin-spiked sediments, one with a TOC of 11% and the other a TOC of 3%. Nebeker et al. (1989) mixed these two sediments to obtain a third sediment with a TOC of 6.1%. Sediments were shaken for 7 days in endrin-coated flasks, and subsequently for 62 days in clean flasks. The 10-day LC50 values for amphipods in the three sediments tested by Nebeker et al. (1989) did not differ when endrin concentration was on a dry weight basis. The LC50 values decreased with increase in organic carbon when the concentration was on an organic carbon basis (Table 4-1). The authors concluded that endrin data do not support equilibrium partitioning theory. LC50 values normalized to dry weight (4.4 to 6.0 μg/g) or wet weight (0.9 to 1.0 μg/g) differed by less than a factor of 1.5 over a 3.7 fold range of TOC. In contrast, the organic carbon–normalized LC50 values ranged from 53.6 to 147 $\mu g/g_{OC}$, a factor of 2.7 (Table 4-1).

Table 4-1. Summary of tests with endrin-spiked sediment

Common Name, *Scientific Name*	Sediment Source	TOC (%)	Method,[a] Duration (days)	Response	Sediment Endrin LC50 Dry wt (μg/g)	Sediment Endrin LC50 OC (μg/g)	Interstitial Water LC50 (μg/L)	Reference
Freshwater Species								
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	1,400	80,000	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	1,050	60,000	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	2,500	143,000	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	5,400	309,000	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	EC50 sediment avoidance	19.0	1,086	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	EC50 sediment avoidance	15.3	874	—	Keilty et al., 1988a
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/54	26% mortality	42.0	2,400	—	Keilty et al., 1988b
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/54	18% mortality	11.5	657	—	Keilty et al., 1988b
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/54	Weight loss	2.33	133	—	Keilty et al., 1988b
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/54	Decreased sediment reworking rate	0.54	30.8	—	Keilty et al., 1988b
Lumbriculid worm, *Stylodrilus heringianus*	Lake Michigan; 0.25mm sieved	1.75[b]	S,N/69	Weight loss	50.0	2,860	—	Keilty and Stehly, 1989
Tubificid worm, *Limnodrilus hoffmeisteri*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	2,050	117,000	—	Keilty et al., 1988a
Tubificid worm, *Limnodrilus hoffmeisteri*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	3,400	194,000	—	Keilty et al., 1988a
Tubificid worm, *Limnodrilus hoffmeisteri*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	LC50	5,600[c]	320,000[c]	—	Keilty et al., 1988a
Tubificid worm, *Limnodrilus hoffmeisteri*	Lake Michigan; 0.25mm sieved	1.75[b]	S,M/4	EC50 sediment avoidance	59.0	3,371	—	Keilty et al., 1988a

Table 4-1. Summary of tests with endrin-spiked sediment (continued)

Common Name, *Scientific Name*	Sediment Source	TOC (%)	Method,[a] Duration (days)	Response	Sediment Endrin LC50 Dry wt (μg/g)	Sediment Endrin LC50 OC (μg/g)	Interstitial Water LC50 (μg/L)	Reference
Amphipod, *Diporeia* sp.	Lake Michigan; depth 29m	0.07	S,M/4	LC50	0.012	17.0	1.07	Stehly, 1992
Amphipod, *Diporeia* sp.	Lake Michigan; depth 45m	0.55	S,M/4	LC50	0.172	31.3	2.2	Stehly, 1992
Amphipod, *Diporeia* sp.	Lake Michigan; depth 100m	1.75	S,M/4	LC50	0.224	12.8	0.63	Stehly, 1992
Amphipod, *Hyalella azteca*	Soap Creek Pond No. 7, OR	3.0	S,M/10	LC50	4.4	147	2.1	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	1:1 mixture of Soap Creek and Mercer Lake, OR	6.1	S,M/10	LC50	4.8	78.7	1.9	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR	11.2	S,M/10	LC50	6.0	53.6	1.8	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	Soap Creek Pond No. 7, OR; refrigerated	3	S,M/10	LC50	5.1	170	—	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Soap Creek Pond No. 7, OR; frozen	3	S,M/10	LC50	7.7	257	—	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; refrigerated	11	S,M/10	LC50	19.6	178	—	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; frozen	11	S,M/10	LC50	21.7	197	—	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; refrigerated	11	S,M/10	LC50	10.3	93.6	—	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; frozen	11	S,M/10	LC50	9.8	89.1	—	Schuytema et al., 1989
Saltwater Species								
Polychaete worm, *Nereis virens*	17% sand, 83% silt and clay[d]	2	R,M/12	2 of 5 worms died	28	1,400	—	McLeese et al., 1982
Sand shrimp, *Crangon septemspinosa*	Sand, wet-sieved between 1-2mm sieves[d]	0.28	R,M/4	LC50	0.047	16.8	—	McLeese and Metcalfe, 1980

[a]S = static, R = renewal, M = measured, N = nominal.
[b]Value from Landrum (1991).
[c]*L. hoffmeisteri* and *S. heringianus* tested together.
[d]Clean sediment placed in endrin-coated beakers at beginning of exposure.

Schuytema et al. (1989) stored an aliquot of sediments dosed by Nebeker et al. (1989) for an average of 9 months and then froze one-half for 2 weeks; the other half was stored at 4°C for 2 weeks. The toxicity of endrin to *H. azteca* did not differ in refrigerated and frozen sediments from Mercer Lake, OR, and differed minimally (LC50 = 5.1 vs 7.7 μg/g dry weight) in sediments from Soap Creek Pond. In contrast to the findings of Nebeker et al. (1989), Schuytema et al. (1989) used the same test sediments and observed higher LC50 values in four tests with Mercer Lake sediments (9.8, 10.3, 19.6, and 21.7 μg/g dry weight), which had a TOC of 11%, than LC50 values from two tests using Soap Creek sediments (5.1 and 7.7 μg/g dry weight) where TOC was 3%.

The only saltwater experiments that tested endrin-spiked sediments were conducted by McLeese et al. (1982) and McLeese and Metcalfe (1980). These began with clean sediments that were added to endrin-coated beakers just before addition of test organisms. This was in marked contrast to tests using freshwater sediments spiked with endrin days or weeks before test initiation (Nebeker et al., 1989; Schuytema et al., 1989). As a result, the endrin concentrations in the sediment and overlying water varied greatly over the course of these experiments. In addition, the transfer of test organisms to freshly prepared beakers every 48 hours adds to the uncertainty associated with the exposure conditions and complicates interpretation of the results of McLeese et al. (1982).

McLeese et al. (1982) tested the effects of endrin on the polychaete worm, *Nereis virens*, in sediment with 2% TOC (17% sand and 83% silt and clay) in 12-day toxicity tests. Only two of five worms died at the highest concentration tested, 28 μg endrin/g dry weight sediment or 1,400 μg endrin/g_{OC}. McLeese and Metcalfe (1980) tested the effects of endrin in sand with a TOC content of 0.28% on the sand shrimp, *Crangon septemspinosa*. The 4-day LC50 was 0.047 μg/g dry weight sediment or 16.8 μg/g_{OC}. Concentrations of endrin in water overlying the sediment were sufficient to explain the observed mortalities of sand shrimp in sediments.

The need for organic carbon normalization of the concentrations of nonionic organic chemicals in sediments is presented in the ESB Technical Basis Document. For endrin, this need is supported by the results of the spiked-sediment toxicity tests described above. When examined individually, experiments in which *H. azteca* were exposed to the same sediments by both Nebeker et al. (1989) and Schuytema et al. (1989) provide contradictory data concerning the need for organic carbon normalization (Table 4-1). Nebeker et al. (1989) observed no change in toxicity with increasing TOC when endrin was expressed on a dry weight basis, whereas Schuytema et al. (1989) observed a decrease in toxicity with increasing TOC when endrin was expressed on a dry weight basis. However, mean LC50 values calculated for individual experiments from both studies were similar when concentrations were normalized by organic carbon content. The mean (geometric) LC50 values were 109 μg/g_{OC} (5 tests) for sediments from Mercer Lake having a TOC of 11% and 186 μg/g_{OC} (3 tests) for sediments from Soap Creek Pond having 3% organic carbon. The lack of consistent evidence supporting organic carbon normalization in the individual tests reported by Nebeker et al. (1989) is in contrast with evidence supporting normalization overall for tests with other nonionic chemicals. The results for sediments spiked with endrin were most likely observed because organic carbon concentrations differed by less than a factor of four and variability inherent in these tests limited the capacity for discrimination. Additional tests by Stehly (1992) provide further support for the need to normalize endrin concentrations in sediments (Table 4-1). The organic carbon concentrations for these sediments ranged from 0.07% to 1.75% (a factor of 25). On a dry weight basis, 4-day LC50 values for *Diporeia* sp. ranged from 0.012 to 0.224 μg/g (a factor of 18.7). The organic carbon–normalized LC50 values were within a factor of 2.4 and ranged from 12.8 to 31.3 μg/g_{OC}.

Although it is important to demonstrate that organic carbon normalization is necessary if ESB_{WQC}s are to be derived using the EqP approach, it is fundamentally more important to demonstrate that K_{OC} and water-only effects concentrations can be used to predict the effects concentration for endrin and other nonionic organic chemicals on an organic carbon basis for a range of sediments. Evidence supporting this prediction for endrin and other nonionic organic chemicals follows in Section 4.3.

4.2 Correlation Between Organism Response and Interstitial Water Concentration

One corollary of the EqP theory is that freely-dissolved interstitial water LC50 values for a given organism should be constant across sediments of varying organic carbon contents (U.S. EPA, 2003a). Appropriate interstitial water LC50 values are available from two studies using endrin (Table 4-1). Nebeker et al. (1989) found 10-day LC50 values for endrin, based on

interstitial water concentrations, ranged from 1.8 to 2.1 μg/L for *H. azteca* exposed to three sediments. Overlying water LC50 values from these static tests (Nebeker et al., 1989) and those conducted using the same sediments by Schuytema et al. (1989) were similar; 1.1 to 3.9 μg/L. Stehly (1992) found that 4-day interstitial water LC50 values for *Diporeia* sp. ranged from 0.63 to 2.2 μg/L (a factor of 3.5); this is considerably less than the range in LC50 values expressed as dry weight, 0.012 to 0.224 μg/g (a factor of 18.7), for three sediments from Lake Michigan having 0.07% to 1.75% organic carbon.

A more detailed evaluation of the degree to which the response of benthic organisms can be predicted from toxic units (TUs) of substances in interstitial water can be made utilizing results from toxicity tests with sediments spiked with a variety of nonionic compounds, including acenaphthene and phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and kepone (Adams et al., 1985) (Figure 4-1). The endrin data included in this analysis were from tests conducted at laboratories or from tests that utilized designs at least as rigorous as those conducted at EPA laboratories. Note that dieldrin data from Hoke et al. (1995) were not used in the interstitial water TU plot either because interstitial water was not measured or because of inconsistencies in the mortality results that have been attributed to DOC complexing in the interstitial water. This is discussed in Hoke et al. (1995) and in the EPA dieldrin ESB document (U.S. EPA, 2003c). Tests with acenaphthene and phenanthrene used two saltwater amphipods (*Leptocheirus plumulosus* and *Eohaustorius estuarius*) and saltwater sediments. Tests with fluoranthene used a saltwater amphipod (*Rhepoxynius abronius*) and saltwater sediments. Freshwater sediments spiked with endrin were tested using the amphipod *H. azteca*, and kepone-spiked sediments were tested using the midge, *C. tentans*.

Figure 4-1 presents the percent mortalities of the benthic species tested in individual treatments for each

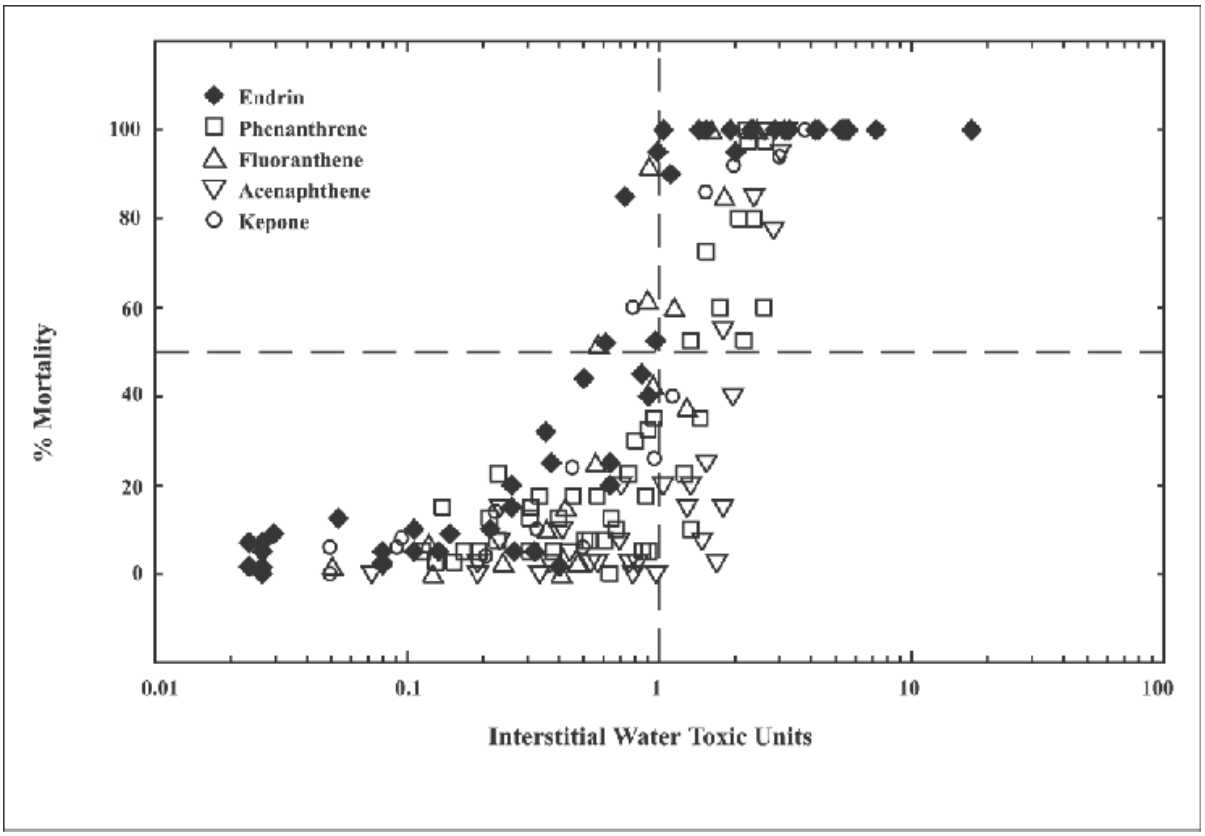


Figure 4-1. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and midge in sediments spiked with kepone (Adams et al., 1985) relative to interstitial water toxic units.

chemical versus interstitial water TUs (IWTUs) for all sediments. IWTUs are the concentration of the chemical in interstitial water (μg/L) divided by the water-only LC50 (μg/L). Theoretically, 50% mortality should occur at 1 IWTU. At concentrations below 1 IWTU, there should be less than 50% mortality, and at concentrations above 1 IWTU there should be greater than 50% mortality. Figure 4-1 shows that, at concentrations below 1 IWTU, mortality was generally low and increased sharply at approximately 1 IWTU. Therefore, this comparison supports the concept that interstitial water concentrations can be used to make a prediction that is not sediment-specific of the response of an organism to a chemical.

4.3 Tests of the Equilibrium Partitioning Prediction of Sediment Toxicity

Sediment benchmarks derived using the EqP approach utilize partition coefficients and FCVs from updated or final WQC documents to derive ESB_{WQC} concentrations that are protective of benthic organisms. The partition coefficient K_{OC} is used to normalize sediment concentrations and predict biologically available concentrations across sediment types. The data required to test the organic carbon normalization for endrin in sediments were available for only one benthic species. Data from tests with water column species were not included in this analysis. Testing of this component of ESB_{WQC} derivation required three elements: (1) a water-only effects concentration, such as a 10-day LC50 value, in μg/L; (2) an identical sediment effect concentration on an organic carbon basis, in $\mu g/g_{OC}$; and (3) a partition coefficient for the chemical, K_{OC}, in L/kg_{OC}. This section presents evidence that the observed effect concentration in sediments (2) can be predicted utilizing the water-only effect concentration (1) and the partition coefficient (3).

Predicted sediment 10-day LC50 values from endrin-spiked sediment tests with *H. azteca* (Nebeker et al., 1989; Schuytema et al., 1989) were calculated (Table 4-2) using the $\log_{10}K_{OC}$ value of 4.97 from Section 2 of this document and the geometric mean of the water-only LC50 value (4.1 μg/L). Overall, ratios of actual to predicted sediment LC50 values for endrin averaged 0.33 (range 0.13 to 0.67) in nine tests with three sediments.

A more detailed evaluation of the accuracy and precision of the EqP prediction of the response of benthic organisms can be made using the results of toxicity tests with amphipods exposed to sediments spiked with acenaphthene, phenanthrene, dieldrin, endrin, or fluoranthene. The data included in this analysis were from tests conducted at EPA laboratories or from tests that utilized designs at least as rigorous as those conducted at EPA laboratories. Data from the kepone experiments were not included because the recommended K_{OW} for kepone obtained from Karickhoff and Long (1995) was evaluated using only one laboratory measured value, whereas the remaining chemical K_{OW} values are recommended based on several laboratory measured values. Swartz (1991) exposed the saltwater amphipods *E. estuarius* and *L. plumulosus* to acenaphthene in three marine sediments having organic carbon contents ranging from 0.82% to 4.2% and to phenanthrene in three marine sediments having organic carbon contents ranging from 0.82% to 3.6%. Swartz et al. (1990) exposed the saltwater amphipod *R. abronius* to fluoranthene in three marine sediments having 0.18%, 0.31%, and 0.48% organic carbon. Hoke et al. (1995) exposed the amphipod *H. azteca* to three dieldrin-spiked freshwater sediments having 1.7%, 2.9%, and 8.7% organic carbon, and also exposed the midge *C. tentans* to two freshwater dieldrin-spiked sediments having 2.0% and 1.5% organic carbon. Nebeker et al. (1989) and Schuytema et al. (1989) exposed *H. azteca* to three endrin-spiked sediments having 3.0%, 6.1%, and 11.2% organic carbon. Figure 4-2 presents the percent mortalities of amphipods in individual treatments of each chemical versus predicted sediment TUs (PSTUs) for each sediment treatment. PSTUs are the concentration of the chemical in sediment ($\mu g/g_{OC}$) divided by the predicted sediment LC50 (i.e., the product of K_{OC} and the 10-day water-only LC50 expressed in $\mu g/g_{OC}$). In this normalization, 50% mortality should occur at 1 PSTU. Figure 4-2 shows that, at concentrations below 1 PSTU, mortality was generally low and increased sharply at 1 PSTU. Therefore, this comparison supports the concept that PSTU values also can be used to make a prediction, that is not sediment-specific, of the response of an organism to a chemical. The means of the LC50 values for these tests, calculated on a PSTU basis, were 1.55 for acenaphthene, 0.73 for dieldrin, 0.33 for endrin, 0.75 for fluoranthene, and 1.19 for phenanthrene. The mean value for the five chemicals was 0.80. The fact that this value is so close to the theoretical value of 1.0 illustrates that the EqP method can account for the effects of different sediment properties and properly predict the effects concentration in sediments using effects concentrations from water-only exposures.

Data variations in Figure 4-2 reflect inherent variability in these experiments and phenomena that

have not been accounted for in the EqP model. The uncertainty of the model is calculated in Section 5.2 of this document. There is an uncertainty of approximately ±2. The error bars shown in Figure 4-2 are computed as ±1.96 × (ESB_{WQC} uncertainty). The value of 1.96 is the t statistic, which provides a 95% confidence interval around the ESB_{WQC}s .

Table 4-2. Water-only and sediment LC50 values used to test the applicability of the EqP theory for endrin

Common Name, *Scientific Name*	Method,[a] Duration (days)	Water-Only LC50 (μg/L)	Overlying Water LC50 (μg/L)	Interstitial Water LC50 (μg/L)	TOC (%)	Endrin Sediment LC50s: Dry Wt. (μg/g)	Endrin Sediment LC50s: OC (μg/g)	Predicted[b] LC50 ($\mu g/g_{OC}$)	Ratio: Actual/ Predicted LC50	Reference
Amphipod, *Hyalella azteca*	S, M/10	4.2[c]	1.3[c]	2.1[c]	3.0	4.4	147	392	0.38	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	3.8[c]	1.1[c]	1.9[c]	6.1	4.8	78.7	355	0.22	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.3[c]	1.2[c]	1.8[c]	11.2	6.0	53.6	401	0.13	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	1.8[c]	—	3	5.1	170	383	0.44	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	3.6[c]	—	3	7.7	257	383	0.67	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	3.6[c]	—	11	19.6	178	383	0.46	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	3.9[c]	—	11	21.7	197	383	0.51	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	1.4[c]	—	11	10.3	93.6	383	0.24	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	S, M/10	4.1[d]	1.8[c]	—	11	9.8	89.1	383	0.23	Schuytema et al., 1989
Geometric Mean		4.1[d]	1.9[c]	1.9[c]	—	—	125.8	383	0.33	

[a]S = static; M = measured.
[b]Predicted LC50 ($\mu g/g_{OC}$) = water-only LC50 (μg/L) × K_{OC} (L/kg_{OC}) × 1 kg_{OC}/1000 g_{OC}; where $K_{OC} = 10^{4.97}$.
[c]Soluble endrin; samples centrifuged prior to analysis.
[d]Mean 10-day water-only LC50 from 3 tests in Nebeker et al. (1989).

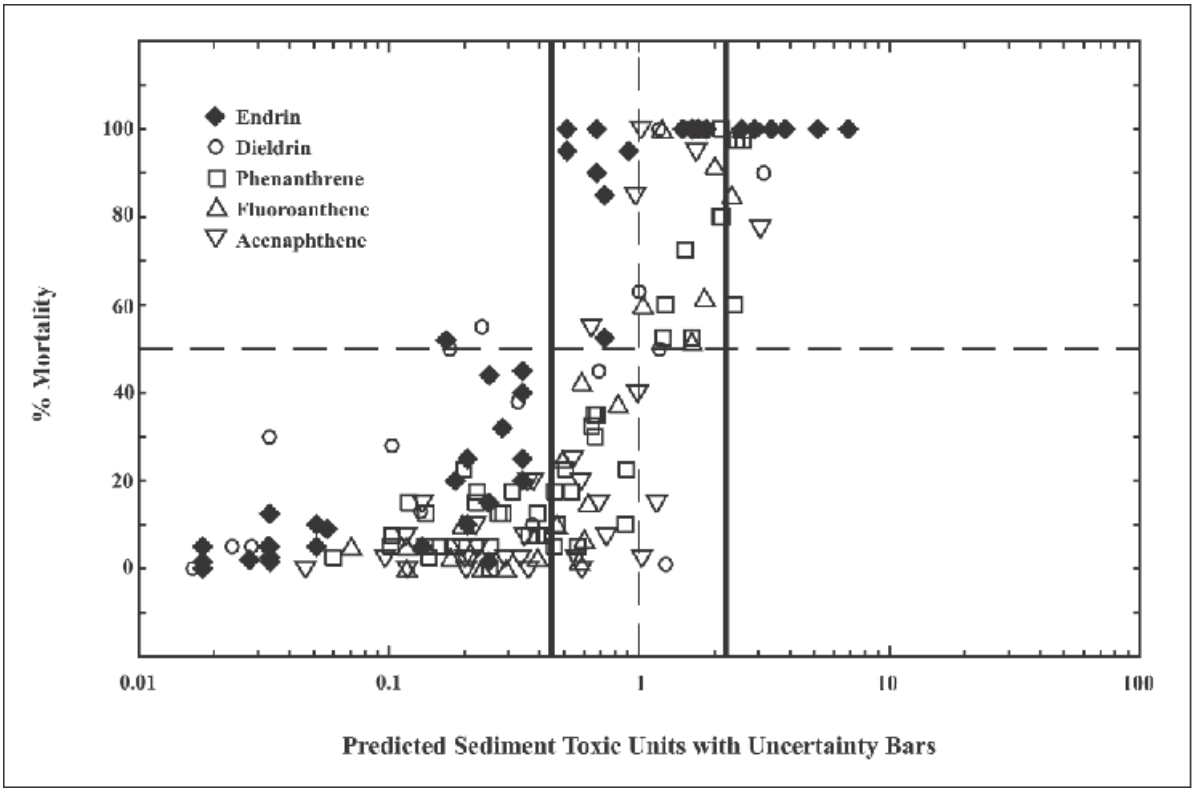


Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke et al., 1995), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and midge in sediments spiked with dieldrin (Hoke et al., 1995) relative to predicted sediment toxic units.

Section 5

Derivation of Endrin ESB_{WQC}s

5.1 Derivation of ESB_{WQC}s

The WQC FCV (see Section 3), without an averaging period or return frequency, can be used to calculate the ESB_{WQC}s because the concentration of contaminants in sediments is probably relatively stable over time. Thus, exposure to sedentary benthic species should be chronic and relatively constant. This contrasts to the situation in the water column, where a rapid change in exposure and exposures of limited durations can occur from fluctuations in effluent concentrations, from dilutions in receiving waters, or from the free-swimming or planktonic nature of water column organisms. For some particular uses of ESB_{WQC}s, it may be appropriate to use the areal extent and vertical stratification of contamination at a sediment site in much the same way that averaging periods or mixing zones are used with WQC.

The FCV is the value that should protect 95% of the tested species included in the calculation of the WQC from chronic effects of the substance. The FCV is the quotient of the FAV and the FACR for the substance. The FAV is an estimate of the acute LC50 or EC50 concentration of the substance corresponding to a cumulative probability of 0.05 from eight or more families for the genera for which acceptable acute tests have been conducted on the substance. The ACR is the mean ratio of acute to chronic toxicity for three or more species exposed to the substance that meets minimum database requirements. For more information on the calculation of ACRs, FAVs, and FCVs, see Section 3 of this document and the National Water Quality Criteria Guidelines (Stephan et al., 1985). The FCV used in this document differs from the FCV in the endrin WQC document (U.S. EPA, 1980) because it incorporates recent data not included in that document and omits some data that do not meet the data requirements established in the 1985 WQC Guidelines.

The EqP method for calculating ESB_{WQC}s are based on the following procedure (also described in Section 2-1). If the FCV (μg/L) is the chronic concentration from the WQC for the chemical of interest, then the ESB_{WQC}s (μg/g sediment) are computed using the partition coefficient, K_P (L/g sediment), between sediment and interstitial water

$$ESB_{WQC}s = K_P \, FCV \qquad (5\text{-}1)$$

The organic carbon partition coefficient, K_{OC}, can be substituted for K_P, because organic carbon is the predominant sorption phase for nonionic organic chemicals in naturally occurring sediments (salinity, grain size, and other sediment parameters have inconsequential roles in sorption; see Sections 2.1 and 4.3). Therefore, on a sediment organic carbon basis, the organic carbon-normalized ESB_{WQC}s (i.e., ESB_{WQCOC} in $\mu g/g_{OC}$) are

$$ESB_{WQCOC} = K_{OC} \, FCV \qquad (5\text{-}2)$$

And because K_{OC} is presumably independent of sediment type for nonionic organic chemicals, so too are the ESB_{WQCOC}. Table 5-1 contains the calculated values of the endrin ESB_{WQC}s.

The ESB_{WQCOC}s are applicable to sediments with $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic

Table 5-1. Equilibrium partitioning sediment benchmarks (ESB_{WQC}s) for endrin using the WQC FCV as the effect concentration

Type of Water Body	Log K_{OW} (L/kg)	Log K_{OC} (L/kg)	FCV (μg/L)	ESB_{WQCOC} ($\mu g/g_{OC}$)
Freshwater	5.06	4.97	0.05805	5.4[a]
Saltwater	5.06	4.97	0.01057	0.99[b]

[a] $ESB_{WQCOC} = (10^{4.97}\ L/kg_{OC}) \times (10^{-3}\ kg_{OC}/g_{OC}) \times (0.05805\ \mu g\ endrin/L) = 5.4\ \mu g\ endrin/g_{OC}$.
[b] $ESB_{WQCOC} = (10^{4.97}\ L/kg_{OC}) \times (10^{-3}\ kg_{OC}/g_{OC}) \times (0.01057\ \mu g\ endrin/L) = 0.99\ \mu g\ endrin/g_{OC}$.

carbon normalization and the resulting ESB_{WQC}s do not apply.

Because organic carbon is the factor controlling the bioavailability of nonionic organic compounds in sediments, ESB_{WQC}s have been developed on an organic carbon basis, not on a dry weight basis. When the chemical concentrations in sediments are reported as dry weight concentrations and organic carbon data are available, it is best to convert the sediment concentrations to μg chemical/g_{OC}. These concentrations can then be directly compared with the ESB_{WQC}s values. This facilitates comparisons between ESB_{WQC}s and field concentrations relative to identification of hot spots and the degree to which sediment concentrations do or do not exceed ESB_{WQC} values. The conversion from dry weight to organic carbon-normalized concentration can be done using the following formula

$$\mu g \text{ chemical}/g_{OC} = \mu g \text{ chemical}/g_{dry\ wt} \div (\% \text{ TOC} \div 100)$$

$$= \mu g \text{ chemical}/g_{dry\ wt} \times 100 \div \% \text{ TOC}$$

For example, a freshwater sediment with a concentration of 0.1 μg endrin/$g_{dry\ wt}$ and 0.5% TOC has an organic carbon–normalized concentration of 20 $\mu g/g_{OC}$ (= 0.1 $\mu g/g_{dry\ wt}$ × 100 ÷ 0.5), which exceeds the freshwater endrin ESB_{WQC} of 5.4 $\mu g/g_{OC}$. Another freshwater sediment with the same concentration of endrin (0.1 $\mu g/g_{dry\ wt}$) but a TOC concentration of 5.0% would have an organic carbon–normalized concentration of 2.0 $\mu g/g_{OC}$ (= 0.1 $\mu g/g_{dry\ wt}$ × 100 ÷ 5.0), which is below the freshwater ESB_{WQC} for endrin.

In situations where TOC values for particular sediments are not available, a range of TOC values may be used in a "worst case" or "best case" analysis. In this case, ESB_{WQCOC} values may be "converted" to dry weight–normalized ESB_{WQC} values ($ESB_{WQCdry\ wt}$). This "conversion" for each level of TOC is

$$ESB_{WQCdry\ wt} = ESB_{WQCOC}\ (\mu g/g_{OC}) \times (\% \text{ TOC} \div 100)$$

For example, the $ESB_{WQCdry\ wt}$ value for freshwater sediments with 1% organic carbon is 0.054 $\mu g/g$

$$ESB_{WQCdry\ wt} = 5.4\ \mu g/g_{OC} \times 1\% \text{ TOC} \div 100 = 0.054\ \mu g/g_{dry\ wt}$$

This method is used in the analysis of the STORET data in Section 5.4.

5.2 Uncertainty Analysis

Some of the uncertainty of the endrin ESB_{WQC}s can be estimated from the degree to which the available sediment toxicity data are predicted using the EqP model, which serves as the basis for the ESBs. In its assertion, the EqP model holds that (1) the bioavailability of nonionic organic chemicals across sediments is equal on an organic carbon basis and (2) the effects concentration in sediment ($\mu g/g_{OC}$) can be estimated from the product of the effects concentrations from water-only exposures (e.g., FCV ($\mu g/L$)) and the partition coefficient, K_{OC} (L/kg). The uncertainty associated with the ESB_{WQC}s can be obtained from a quantitative estimate of the degree to which the available data support these assertions.

The data used in the uncertainty analysis are from the water-only and sediment toxicity tests that were conducted to fulfill the minimum database requirements for development of the ESB_{WQC}s (see Section 4.3 and the ESB Technical Basis Document). These freshwater and saltwater tests span a range of chemicals and organisms, they include exposures using water-only and a number of sediments and are replicated within each chemical-organism-exposure media treatment. These data are analyzed using an analysis of variance (ANOVA) to estimate the uncertainty (i.e., the variance) associated with the varying exposure media and that associated with experimental error. If the EqP model were perfect, then there would be experimental error only. Therefore, the uncertainty associated with the use of EqP is the variance associated with varying exposure media.

The data used in the uncertainty analysis are illustrated in Figure 4-2. The data for endrin are summarized in Appendix B. LC50 values for sediment and water-only tests were computed from these data. The EqP model can be used to normalize the data in order to put it on a common basis. The LC50 values from water-only exposures ($LC50_W$; $\mu g/L$) are related to the organic carbon–normalized LC50 values from sediment exposures ($LC50_{S,OC}$; $\mu g/g_{OC}$) via the partitioning equation

$$LC50_{S,OC} = K_{OC} LC50_W \qquad (5\text{-}3)$$

As mentioned above, one of the assertions of the EqP model is that the toxicity of sediments expressed on an

organic carbon basis equals the toxicity in water tests multiplied by the K_{OC}. Therefore, both $LC50_{S,OC}$ and $K_{OC} \times LC50_W$ are estimates of the true $LC50_{OC}$ for each chemical–organism pair. In this analysis, the uncertainty of K_{OC} is not treated separately. Any error associated with K_{OC} will be reflected in the uncertainty attributed to varying the exposure media.

In order to perform an analysis of variance, a model of the random variations is required. As discussed above, experiments that seek to validate Equation 5-3 are subject to various sources of random variations. A number of chemicals and organisms have been tested. Each chemical–organism pair was tested in water-only exposures and in different sediments. Let α represent the random variation due to this source. Also, each experiment was replicated. Let ϵ represent the random variation due to this source. If the model were perfect, there would be no random variations other than those from experimental error, which is reflected in the replications. Hence, α represents the uncertainty due to the approximations inherent in the model and ϵ represents the experimental error. Let $(\sigma_\alpha)^2$ and $(\sigma_\epsilon)^2$ be the variances of these random variables. Let i index a specific chemical–organism pair. Let j index the exposure media, water-only, or the individual sediments. Let k index the replication of the experiment. Then the equation that describes this relationship is

$$\ln(LC50_{i,j,k}) = \mu_i + \alpha_{i,j} + \epsilon_{i,j,k} \qquad (5\text{-}4)$$

where $\ln(LC50_{i,j,k})$ is either $\ln(LC50_W)$ or $\ln(LC50_{S,OC})$, corresponding to a water-only or sediment exposure, and μ_i is the population ln(LC50) for chemical-organism pair i. The error structure is assumed to be log normal which corresponds to assuming that the errors are proportional to the means (e.g., 20%), rather than absolute quantities (e.g., 1 $\mu g/g_{OC}$). The statistical problem is to estimate μ_i, $(\sigma_\alpha)^2$, and $(\sigma_\epsilon)^2$. The maximum likelihood method is used to make these estimates (U.S. EPA, 2003a). The results are shown in Table 5-2. The last line of Table 5-2 is the uncertainty associated with the ESB_{WQC}s; i.e., the variance associated with the exposure media variability.

The confidence limits for the ESB_{WQC}s are computed using this estimate of uncertainty for the ESB_{WQC}s. For the 95% confidence interval limits, the significance level is 1.96 for normally distributed errors.
Hence,

$$\ln(ESB_{WQCOC})_{UPPER} = \ln(ESB_{WQCOC}) + 1.96\sigma_{ESBWQC} \qquad (5\text{-}5)$$

$$\ln(ESB_{WQCOC})_{LOWER} = \ln(ESB_{WQCOC}) - 1.96\sigma_{ESBWQC} \qquad (5\text{-}6)$$

The confidence limits are given in Table 5-3.

The ESB_{WQCOC}s are applicable to sediments with $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic

Table 5-2. Analysis of variance for derivation of confidence limits of the ESB_{WQC}s for endrin

Source of Uncertainty	Parameter	Value ($\mu g/g_{OC}$)
Exposure media	σ_α	0.41
Replication	σ_ϵ	0.29
ESB_{WQC} sediment benchmark	$\sigma_{ESB_{WQC}}$[a]	0.41

[a] $\sigma_{ESB} = \sigma_\alpha$.

Table 5-3. Confidence limits of the ESB_{WQC}s for endrin

Type of Water Body	ESB_{WQCOC} ($\mu g/g_{OC}$)	95% Confidence Limits ($\mu g/g_{OC}$)	
		Lower	Upper
Freshwater	5.4	2.4	12
Saltwater	0.99	0.44	2.2

carbon normalization and ESB_{WQC}s do not apply.

5.3 Comparison of Endrin ESB_{WQC}s and Uncertainty Concentrations to Sediment Concentrations that are Toxic or Predicted to be Chronically Acceptable

Insight into the magnitude of protection afforded to benthic species by ESB_{WQC} concentrations and 95% confidence intervals can be inferred using effect concentrations from toxicity tests with benthic species exposed to sediments spiked with endrin and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figures 5-1 and 5-2). The effect concentrations are predicted from water-only toxicity data and K_{OC} values (see

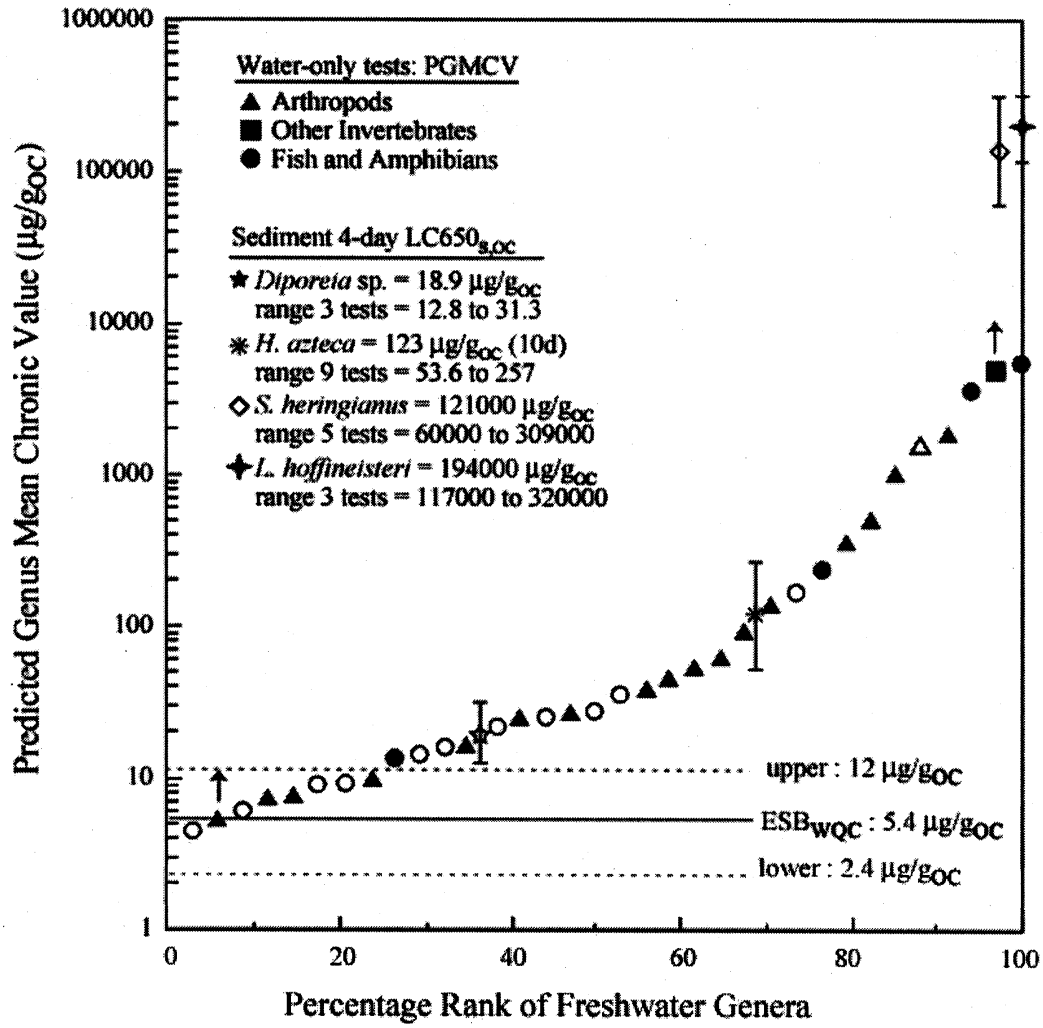


Figure 5-1. **Predicted genus mean chronic values (PGMCV)calculated from water-only toxicity values (Equation 5-7; Appendix A) using freshwater species versus percentage rank of their sensitivity. Lines indicate the freshwater endrin ESB_{WQC} ± 95% confidence limits. Solid symbols are benthic genera; open symbols are water column ge nera. Arrows indicate greater than values. Sediment 4-day $LC50_{s,OC}$ values (calculated from Keilty et al., 1988a; and Keilty and Stehly, 1989; Nebeker et al. 1989; Schuytema et al., 1989; Stehly, 1992; see Table 4-1) for the amphipods *Diporeia* sp. (★) and *H. azteca* (✳) and lumbriculid worm (*S. heringianus*; ◊) and tubificid worm (*L. hoffmeisteri*; ✦) are provided for comparison. Error bars around sediment $LC50_{s,OC}$ values indicate observed range of LC50s.**

Section 4). Chronically acceptable concentrations are extrapolated from GMAVs from water-only, 96-hour lethality tests using the FACR. These two predictive values are used to estimate chronically acceptable sediment concentrations (predicted genus mean chronic values, PGMCV) for endrin from GMAVs (Appendix A), the FACR (Table 3-2), and the K_{OC} (Table 5-1)

$$PGMCV = (GMAV \div ACR)K_{OC} \quad (5\text{-}7)$$

Each PGMCV for fishes and amphibians, arthropods or other invertebrates tested in water was plotted against the percentage rank of its sensitivity. Results from toxicity tests with benthic organisms exposed to sediments spiked with endrin (Table 4-1; Appendix B) were placed in the PGMCV rank appropriate to the test-specific effect concentration. For example, the mean 10-day $LC50_{S,OC}$ for *H. azteca*, 126 $\mu g/g_{OC}$, was placed between the PGMCV of 92 $\mu g/g_{OC}$ for the amphipod, *Gammarus*, and the PGMCV of 138 $\mu g/g_{OC}$ for the dipteran, *Atherix*. Therefore, the LC50 or other effect concentrations are intermingled in this figure with concentrations predicted to be chronically safe. Care should be taken by the reader in interpreting these data with dissimilar endpoints. The following discussion of ESB_{WQC}s,

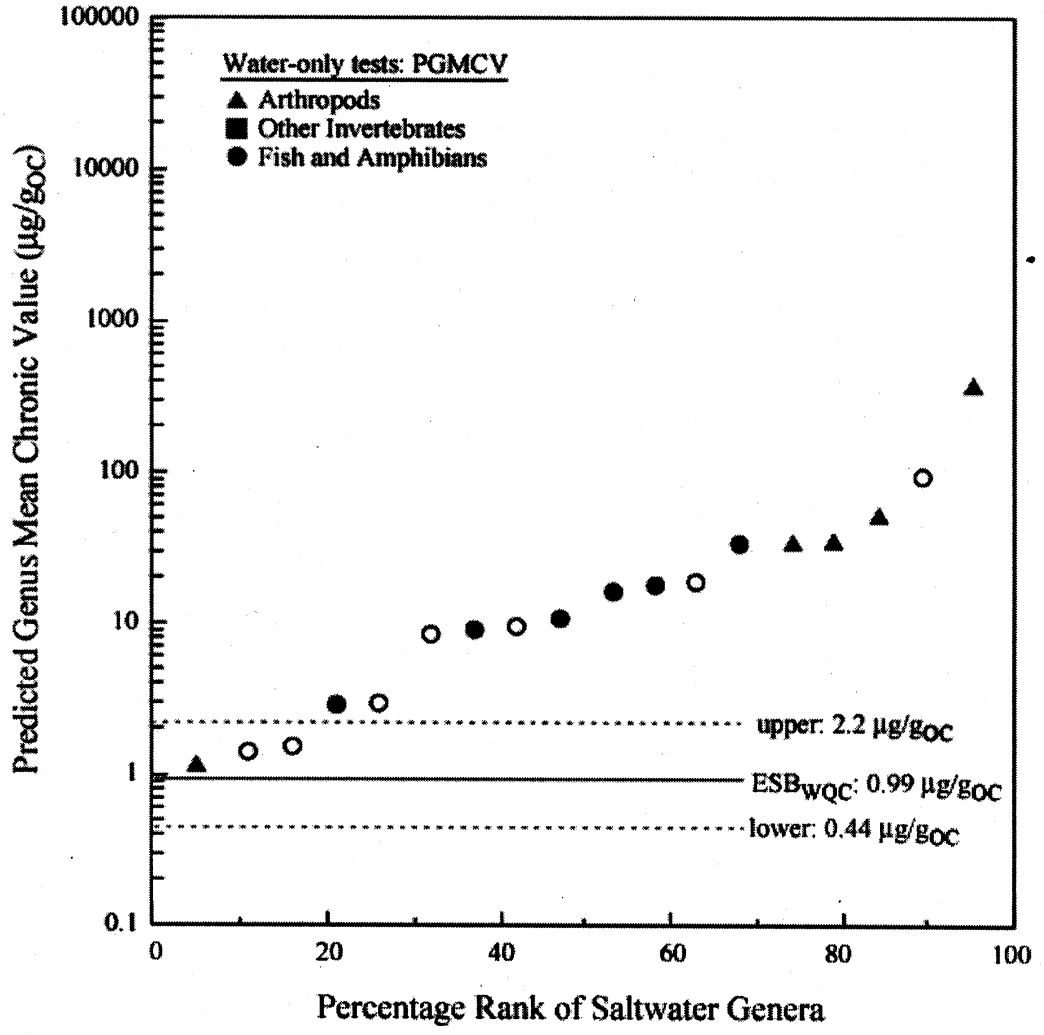


Figure 5-2. Predicted genus mean chronic values (PGMCV) calculated from water-only toxicity values (Equation 5-7; Appendix A) using saltwater species versus percentage rank of their sensitivity. Solid symbols are benthic genera; open symbols are water column genera.

organism sensitivities, and PGMCVs is not intended to provide accurate predictions of the responses of taxa or communities of benthic organisms relative to specific concentrations of endrin in sediments in the field. It is, however, intended to guide scientists and managers through the complexity of available data relative to potential risks to benthic taxa posed by sediments contaminated with endrin.

Figures 5-1 and 5-2 are recreations of Figures 3-1 and 3-2, respectively, with GMAVs taken from Appendix A to calculate PGMCVs using Equation 5-7. The freshwater ESB_{WQC} for endrin (5.4 $\mu g/g_{OC}$) is less than 33 of the 34 PGMCVs and all of the LC50 values from spiked-sediment toxicity tests (Figure 5-1). The PGMCV for the fish *Perca* (4.5 $\mu g/g_{OC}$) is less than the ESB_{WQC}. PGMCVs for 26 of 34 freshwater genera are greater than the upper 95% confidence interval of the ESB_{WQC}(12 $\mu g/g_{OC}$). The PGMCVs for eight genera, including four water column fish and four benthic arthropod genera, are below the ESB_{WQC} upper 95% confidence interval. This illustrates why the slope of the species sensitivity distribution is important. It also suggests that if the extrapolation from water-only acute lethality tests to chronically acceptable sediment concentrations is accurate, these or similarly sensitive genera may be chronically affected by sediment concentrations marginally less than the ESB_{WQC} and possibly less than the 95% upper confidence interval. For endrin, the PGMCVs ranged over three orders of magnitude from the most sensitive to the most tolerant genus. A sediment concentration 10 times the ESB_{WQC} would exceed the PGMCVs of 10 of the 22 benthic genera tested including stoneflies, caddisflies, mayflies, dipterans, isopods, and fish. Tolerant benthic genera such as the annelid *Lumbriculus* may not be chronically affected in sediments with endrin concentrations almost 1,000 times the ESB_{WQC}. Data from lethality tests with two freshwater amphipods and two freshwater annelids exposed to endrin-spiked sediments substantiate this range of sensitivity. The LC50 values from these tests range from 2.4 to 59,000 times the ESB_{WQC} of 5.4 $\mu g/g_{OC}$.

The saltwater ESB_{WQC} for endrin (0.99 $\mu g/g_{OC}$) is less than any of the PGMCVs for saltwater genera (Figure 5-2). The PGMCVs for the penaeid shrimp *Penaeus* (1.1 $\mu g/g_{OC}$) and the fishes *Oncorhynchus* (1.44 $\mu g/g_{OC}$) and *Menidia* (1.50 $\mu g/g_{OC}$) are lower than the upper 95% confidence interval for the ESB_{WQC} (2.2 $\mu g/g_{OC}$). For endrin, PGMCVs from the most sensitive to the most tolerant saltwater genus range over two orders of magnitude. A sediment concentration 20 times the ESB_{WQC} would exceed the PGMCVs of 6 of the 11 benthic genera tested including 1 arthropod and 5 fish genera. The hermit crab *Pagurus* is less sensitive and might not be expected to be chronically affected in sediments with endrin concentrations 300 times the ESB_{WQC}.

5.4 Comparison of Endrin ESB_{WQC}sto STORET and Corps of Engineers, San Francisco Bay Databases for Sediment Endrin

Endrin is frequently measured when samples are taken to measure sediment contamination, and endrin values are frequently reported in databases of sediment contamination. This means that it is possible that many of the sediments from the nation's waterways might exceed the endrin benchmarks. In order to investigate this possibility, the endrin benchmarks were compared with data from several available databases of sediment chemistry.

The following description of endrin distributions in Figure 5-3 is somewhat misleading because it includes data from most samples in which the endrin concentration was below the detection limit. These data are indicated on the plot as "less than" symbols (<), but are plotted at the reported detection limits. Because these values represent upper bounds, not measured values, the percentage of samples in which the ESB_{WQC} values were actually exceeded may be less than the reported percentage. Very few of the measured values from either of the databases exceeded the ESB_{WQC}s.

A STORET (U.S. EPA, 1989b) data retrieval was performed to obtain a preliminary assessment of the concentrations of endrin in the sediments of the nation's water bodies. Log probability plots of endrin concentrations on a dry weight basis in sediments are shown in Figure 5-3. Endrin was found at significant concentrations in sediments from rivers, lakes, and near-coastal water bodies in the United States. This is because of its widespread use and the quantity applied during the 1970s and 1980s. It was banned on October 10, 1984. Median concentrations were generally at or near detection limits in most water bodies. There is significant variability in endrin concentrations in sediments throughout the country. Lake samples in EPA Region 9 appear to have had relatively high endrin levels (median = 0.030 $\mu g/g$) prior to 1986. The upper 10% of the concentrations were disproportionally found in streams, rivers, and

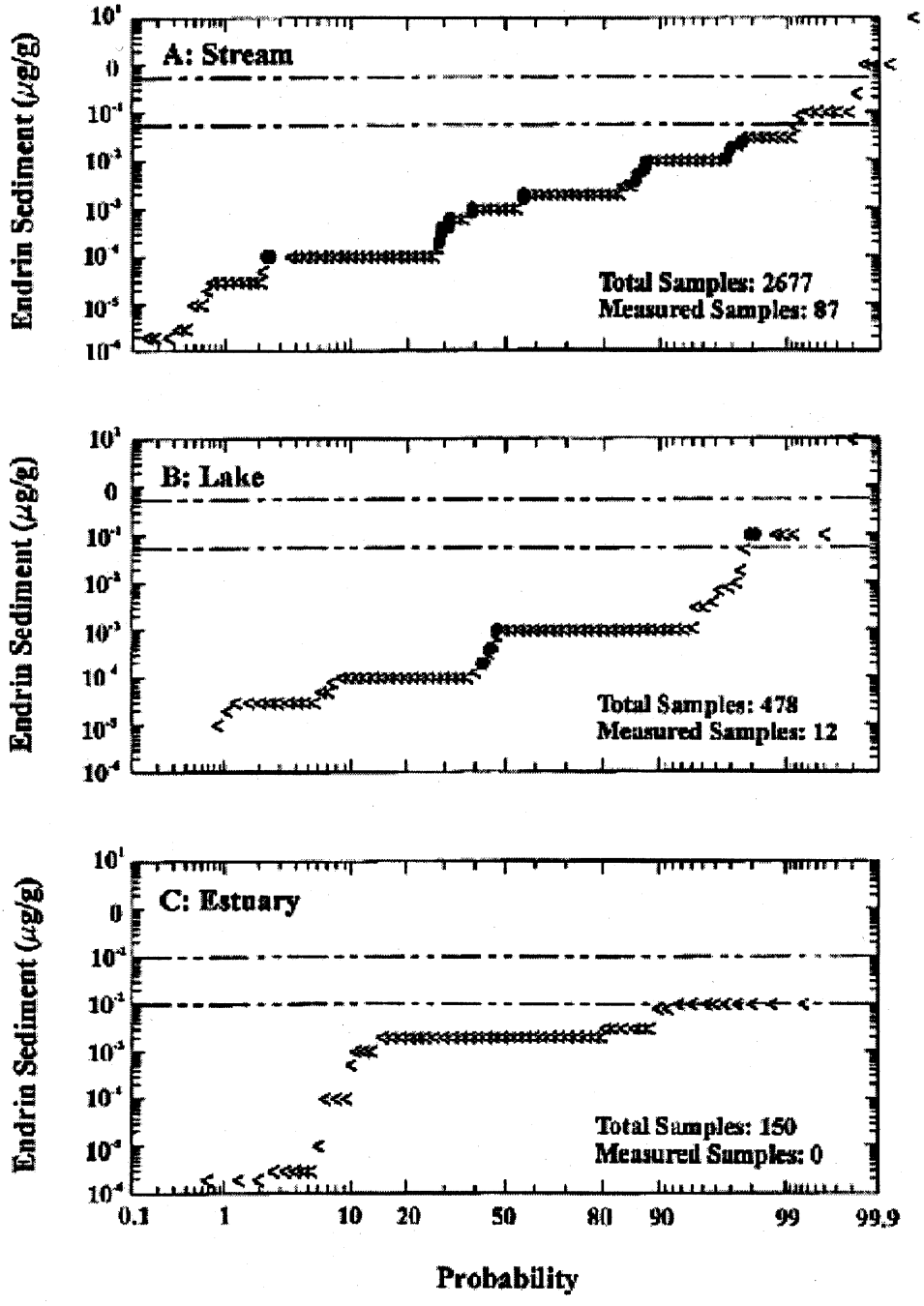


Figure 5-3. Probability distribution of concentrations of endrin in sediments from streams (A), lakes (B), and estuaries (C) in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared with the endrin ESB_{WQC} values. Sediment endrin concentrations below the detection limits are shown as less than symbols (<); measured concentrations are shown as solid circles (•). The upper dashed line on each figure represents the ESB_{WQC} value when TOC=10%, the lower dashed line represents the ESB_{WQC} when TOC=1%.

lakes in EPA Region 7 and in streams, rivers, lakes, and estuaries in Region 9 prior to 1986. In some streams and rivers in Region 7, concentrations remained high after 1986 (Figure 5-3).

The ESB_{WQC}s for endrin can be compared to existing concentrations of endrin in sediments of natural water systems in the United States as contained in the STORET database (U.S. EPA, 1989b). These data were generally reported on a dry weight basis rather than an organic carbon–normalized basis. Therefore, ESB_{WQC} concentrations corresponding to sediment organic carbon levels of 1% to 10% were compared with endrin's distribution in sediments as examples only. For freshwater sediments, ESB_{WQC} concentrations were 0.054 μg/g dry weight in sediments having 1% organic carbon and 0.54 μg/g dry weight in sediments having 10% organic carbon; for marine sediments, the ESB_{WQC}s were 0.0099 μg/g dry weight and 0.099 μg/g dry weight, respectively. Figure 5-3 presents the comparisons of these ESB_{WQC}s with probability distributions of observed sediment endrin levels for streams and lakes (freshwater systems, A and B) and estuaries (marine systems, C).

For streams (n = 2,677), the ESB_{WQC}s of 0.054 μg/g dry weight for 1% organic carbon sediments and 0.54 μg/g dry weight for 10% organic carbon freshwater sediments were exceeded in less than 1% of the samples. For lakes (n = 478), the ESB_{WQC} of 0.054 μg/g dry weight for 1% organic carbon sediment was exceeded in about 2% of the samples, and the ESB_{WQC} of 0.54 μg/g dry weight for 10% organic carbon freshwater sediments was exceeded in less than 1% of the samples. In estuaries, the data (n = 150) indicate that the ESB_{WQC} of 0.0099 μg/g dry weight sediment for 1% organic carbon sediments was exceeded in about 8% of the samples, and the ESB_{WQC} of 0.099 μg/g dry weight for 10% organic carbon freshwater sediments was not exceeded by any of the samples.

A second set of data was analyzed, from the U.S. Army Corps of Engineers (1991) monitoring program, for a number of locations in various parts of San Francisco Bay. For a listing of locations sampled, the number of observations at each site, and the period during which the results were obtained, see U.S.EPA (2003a). These data were collected to examine the

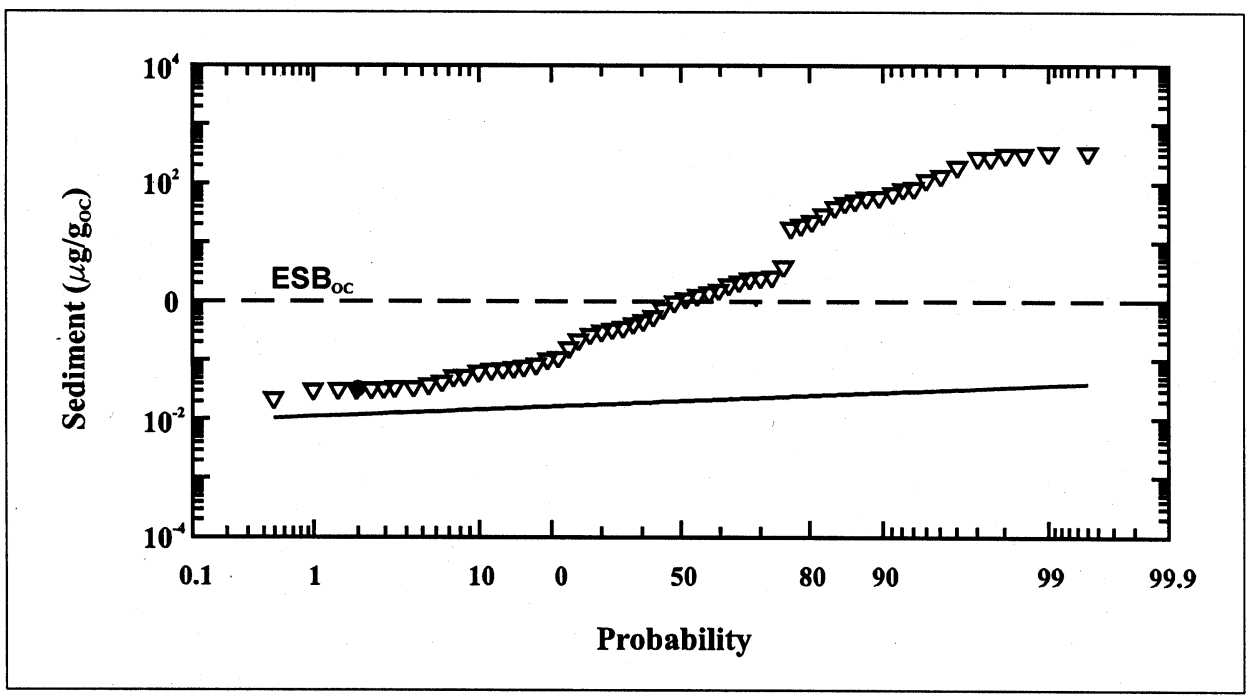


Figure 5-4. Probability distribution of organic carbon–normalized sediment endrin concentrations from the U.S. Army Corps of Engineers (1991) monitoring program of San Francisco Bay. Sediment endrin concentrations less than the detection limits are shown as open triangles (▽); measured concentrations are shown as solid circles (•). The solid line is an estimate of the distribution developed by accounting for nondetected observations.

quality of dredged sediments in order to determine their suitability for open water disposal. The database did not indicate what determinations were made concerning their acceptability for this purpose.

Investigators compared the frequency of occurrence of a given sediment endrin concentration (in individual samples, not dredge sites) with the ESB_{WQC}s developed using the EqP methodology. A major portion (93%) of the samples analyzed had f_{OC} >0.2%, for which the ESB_{WQC} concentrations are applicable. The concentrations of endrin measured in sediments were normalized by the organic carbon content, and the results are displayed as a probability plot in Figure 5-4 to illustrate the frequency at which different levels are observed. Nearly all of the samples were less than the varying detection limits of the analytical tests. Each of the samples for which an actual measurement was obtained was at least an order of magnitude lower than the ESB_{WQC}s. An estimate of the possible frequency distribution of sediment concentrations of endrin was developed by the application of an analysis technique that accounts for the varying detection limits and the presence of nondetected observations (El-Sharrawi and Dolan, 1989). The results are illustrated by the straight line, which suggests that no appreciable number of exceedences is expected. However, the virtual absence of detected concentrations makes the distribution estimates unreliable. They are presented only to suggest the probable relationship between the levels of these two pesticides in relation to sediment benchmarks.

Regional-specific differences in endrin concentrations may affect the above conclusions concerning expected example benchmarks exceedences. This analysis also does not consider other factors such as the type of samples collected (i.e., whether samples were from surficial grab samples or vertical core profiles) or the relative frequencies and intensities of sampling in different study areas. It is presented as an aid in assessing the range of reported endrin sediment concentrations and the extent to which they may exceed the ESB_{WQC}s.

5.5 Limitations to the Applicability of ESBs

Rarely, if ever, are contaminants found alone in naturally occurring sediments. Obviously, the fact that the concentration of a particular contaminant does not exceed the ESBs does not mean that other chemicals, for which there are no ESBs available, are not present in concentrations sufficient to cause harmful effects. Furthermore, even if ESBs were available for all of the contaminants in a particular sediment, there might be additive or synergistic effects that the benchmarks do not address. In this sense, the ESBs represent a "best case" benchmark.

It is theoretically possible that antagonistic reactions between chemicals could reduce the toxicity of a given chemical such that it might not cause unacceptable effects on benthic organisms at concentrations above the ESBs when it occurs with the antagonistic chemical. However, antagonism has rarely been demonstrated. More common would be instances where toxic effects occur at concentrations below the ESBs because of the additive toxicity of many common contaminants such as heavy metals and polycyclic aromatic hydrocarbons (PAHs) (Alabaster and Lloyd, 1982), and instances where other toxic compounds for which no ESBs exist occur along with ESB chemicals.

Care must be used in the application of benchmarks in disequilibrium conditions. In some instances, site-specific ESBs may be required to address disequilibrium. The ESBs assume that nonionic organic chemicals are in equilibrium with the sediment and interstitial water and are associated with sediment primarily through adsorption to sediment organic carbon. In order for these assumptions to be valid, the chemical must be dissolved in interstitial water and partitioned into sediment organic carbon. Therefore, the chemical must be associated with the sediment for a sufficient length of time for equilibrium to be reached. In sediments where particles of undissolved endrin occur, disequilibrium exists and the benchmarks are overprotective. In liquid chemical spill situations, disequilibrium concentrations in interstitial and overlying water may be proportionately higher relative to sediment concentrations. In this case, the benchmarks may be underprotective.

Note that the K_{OC} values used in the EqP calculations described in this document assume that the organic carbon in sediments is similar in partitioning properties to "natural" organic carbon found in most sediments. While this has proven true for most sediments EPA has studied, it is possible that some sites may have components of sediment organic carbon with different properties. This might be associated with sediments whose composition has been highly modified by industrial activity, resulting in high

percentages of atypical organic carbon such as rubber, animal processing waste (e.g., hair or hide fragments), coal particles, or wood processing wastes (bark, wood fiber, or chips). Relatively undegraded woody debris or plant matter (e.g., roots, leaves) may also contribute organic carbon that partitions differently from typical organic carbon (e.g., Iglesias-Jimenez et al., 1997; Grathwohl, 1990; Xing et al., 1994). Sediments with substantial amounts of these materials may exhibit higher concentrations of chemicals in interstitial water than would be predicted using generic K_{OC} values, thereby making the ESBs underprotective. If such a situation is encountered, the applicability of literature K_{OC} values can be evaluated by analyzing for the chemical of interest in both sediment and interstitial water. If the measured concentration in interstitial water is markedly greater (e.g., more than twofold) than that predicted using the K_{OC} values recommended herein (after accounting for DOC binding in the interstitial water), then the ESBs would be underprotective and calculation of a site-specific ESB should be considered (see U.S. EPA, 2003b).

The presence of organic carbon in large particles may also influence the apparent partitioning. Large particles may artificially inflate the effect of the organic carbon because of their large mass, but comparatively small surface area; they may also increase variability in TOC measurements by causing sample heterogeneity. The effect of these particles on partitioning can be evaluated by analysis of interstitial water as described above, and site-specific ESBs may be used if required. It may be possible to screen large particles from sediment prior to analysis to reduce their influence on the interpretation of sediment chemistry relative to ESBs.

In very dynamic areas, with highly erosional or depositional bedded sediments, equilibrium may not be attained with contaminants. However, even high K_{OW} nonionic organic compounds come to equilibrium in clean sediment in a period of days, weeks, or months. Equilibrium times are shorter for mixtures of two sediments that each have previously been at equilibrium. This is particularly relevant in tidal situations where large volumes of sediments are eroded and deposited, even though near equilibrium conditions may predominate over large areas. Except for spills and particulate chemical, near equilibrium is the rule and disequilibrium is less common. In instances where it is suspected that EqP does not apply for a particular sediment because of the disequilibrium discussed above, site-specific methodologies may be applied (U.S. EPA, 2003b). .

Section 6

Sediment Benchmark Values: Application and Interpretation

6.1 Benchmarks

Based on the level of protection provided by WQC, the procedures described in this document indicate that benthic organisms should be comparably protected from adverse effects of endrin where endrin concentrations in sediment are below the ESB_{WQC} values of 5.4 µg endrin/g_{OC} for freshwater sediments and 0.99 µg endrin/g_{OC} for marine/estuarine sediments, except possibly where a locally important species is very sensitive or sediment organic carbon is $< 0.2\%$.

Confidence limits of 2.4 to 12 µg/g_{OC} for freshwater sediments and 0.44 to 2.2 µg/g_{OC} for marine/estuarine sediments are provided as an estimate of the uncertainty associated with the degree to which toxicity can be predicted using the K_{OC} and the water-only effects concentration. Confidence limits do not incorporate uncertainty associated with water quality criteria, or unusual, site-specific circumstances. An understanding of the theoretical basis of the equilibrium partitioning methodology, uncertainty, and the partitioning and toxicity of endrin are required in the use of ESBs and their confidence limits.

The benchmarks presented in this document are the concentrations of a substance that may be present in sediment while still protecting benthic organisms from the effects of that substance. These benchmarks are applicable to a variety of freshwater and marine sediments because they are based on the biologically available concentration of the substance in those sediments.

These benchmarks do not protect against additive, synergistic, or antagonistic effects of contaminants or bioaccumulative effects to aquatic life, wildlife or human health. Consistent with the recommendations of EPA's Science Advisory Board, publication of these documents does not imply the use of ESBs as stand-alone, pass-fail criteria for all applications; rather, exceedances of ESBs could trigger collection of additional assessment data.

6.2 Considerations in the Application and Interpretation of ESBs (also see Section 5.5)

6.2.1 *Relationship of ESB_{WQC} to Expected Effects*

The ESB_{WQC} should be interpreted as a chemical concentration below which adverse effects are not expected. In comparison, at concentrations above the ESB_{WQC} effects may occur. In principle, above the upper confidence limit effects are expected if the chemical is bioavailable as predicted by EqP theory. In general terms, the degree of effect expected increases with increasing endrin concentration in the sediment. Because the FCV is derived as an estimate of the concentration causing chronic toxicity to sensitive organisms, effects of this type may be expected when sediment concentrations are near the ESB_{WQC}. As sediment concentrations increase beyond the ESB_{WQC}, one can expect chronic effects on less sensitive species and/or acute effects on sensitive species.

6.2.2 *Use of EqP to Develop Alternative Benchmarks*

The FCV is used to define a threshold for unacceptable effects based on its precedence in establishing unacceptable effects in the development of WQC. However, the use of EqP to assess sediment contamination is not limited to the ESB_{WQC} and the associated level of protection. By substituting water-only effect values other than the FCV into the ESB equation, other benchmarks may be developed that are useful in evaluating specific types of biological effects, or that better represent the ecological protection goals for specific assessments.

6.2.3 *Influence of Unusual Forms of Sediment Organic Carbon*

Partition coefficients used for calculating these ESBs are based on estimated and measured partitioning from natural organic carbon in typical field sediments. Some sediments influenced heavily by anthropogenic activity may contain sources of organic carbon whose partitioning properties are not similar, such as rubber, animal processing wastes (e.g., hair or hide fragments), or wood processing wastes (bark, wood fiber or chips). Relatively undegraded woody debris or plant matter (e.g., roots, leaves) may also contribute organic carbon that results in partitioning different from that of typical organic carbon. Sediments with large amounts of these materials may show higher concentrations of chemicals in interstitial water than would be predicted using generic K_{OC} values, making the ESBs underprotective. Direct analysis of interstitial water can be used to evaluate this possibility (see U.S. EPA, 2003a,b); if necessary, derivation of site-specific K_{OC} values may be warranted.

6.2.4 *Relationship to Risks Mediated through Bioaccumulation and Trophic Transfer*

As indicated above, ESBs are designed to address direct toxicity to benthic organisms exposed directly to contaminated sediment. They are not designed to address risks that may occur through bioaccumulation and subsequent exposure of pelagic aquatic organisms (e.g., predatory fish), terrestrial or avian wildlife, or humans. No inference can be drawn between attainment of the ESB_{WQC} and the potential for risk via bioaccumulation; the potential for those risks must be addressed by separate means.

6.2.5 *Exposures to Chemical Mixtures*

The methodology described in this document can be used to derive ESB_{WQC}s that protect against the specific toxic effects of endrin; it does not account for potential antagonistic, additive, or synergistic effects that may occur in sediments containing a mixture of endrin and other chemicals. Consideration of this potential must be on a site-specific basis. In general terms, it might be expected that chemicals with toxicological modes of action similar to endrin may show additive toxicity with endrin

6.2.6 *Interpreting ESBs in Combination with Toxicity Tests*

Sediment toxicity tests provide an important complement to ESBs in interpreting overall risk from contaminated sediments. Toxicity tests have different strengths and weaknesses compared to chemical-specific guidelines, and the most powerful inferences can be drawn when both are used together.

Unlike chemical-specific guidelines, toxicity tests are capable of detecting any toxic chemical, if it is present in toxic amounts; one does not need to know what the chemicals of concern are to monitor the sediment. Toxicity tests are also useful for detecting the combined effect of chemical mixtures, if those effects are not considered in the formulation of the applicable chemical-specific guideline.

On the other hand, toxicity tests have weaknesses also; they provide information only for the species tested, and also only for the endpoints measured. This is particularly critical given that most sediment toxicity tests conducted at the time of this writing primarily measure short-term lethality; chronic test procedures have been developed and published for some species, but these procedures are more resource-intensive and have not yet seen widespread use. In contrast, the ESB_{WQC} is intended to protect most species against both acute and chronic effects.

Many assessments may involve comparison of sediment chemistry (relative to ESBs or other sediment quality guidelines) and toxicity test results. In cases where results using these two methods agree (either both positive or both negative), the interpretation is clear. In cases where the two disagree, the interpretation is more complex and required further evaluation.

Individual ESBs address only the effects of the chemical or group of chemicals for which they are derived. For this reason, if a sediment shows toxicity but does not exceed the ESB_{WQC} for a chemical of interest, it is likely that the cause of toxicity is a different chemical or chemicals. This result might

also occur if the partitioning of the chemical in a sediment is different from that assumed by the K_{OC} value used (see "6.2.3 Influence of Unusual Forms of Sediment Organic Carbon" above).

In other instances, it may be that an ESB_{WQC} is exceeded but the sediment is not toxic. As explained above, these findings are not mutually exclusive, because the inherent sensitivity of the two measures is different. The ESB_{WQC} is intended to protect relatively sensitive species against both acute and chronic effects, whereas toxicity tests are run with specific species that may or may not be sensitive to chemicals of concern, and often do not encompass the most sensitive endpoints (e.g., growth or reproduction). As such, one would not expect an endrin concentration near the ESB_{WQC} to cause lethality in a short-term test. It is also possible for a sediment above the ESB_{WQC} to be non-toxic if there are site-specific conditions that run counter to the equilibrium partitioning model and its assumptions as outlined in this document.

A good method for evaluating the results of toxicity tests is to calculate effect concentrations in sediment that are species and endpoint specific. For species contained in the water-only toxicity data for the endrin ESB_{WQC}s (Section 3), effect concentrations in sediment can be calculated that are specific for that organism using procedures in Section 5. These values could then be used to directly judge whether the absence of toxicity in the toxicity test would be expected from the concentration of endrin present.

If the exceedance of an ESB is sufficient that one would expect effects in a toxicity test but they are not observed, it is prudent to evaluate the partitioning behavior of the chemical in the sediment. This is performed by isolating interstitial water from the sediment and analyzing it for endrin. Predicted concentrations of endrin in the interstitial water can be calculated from the measured concentrations in the solid phase (normalized to organic carbon) as follows

$$\mu g \text{ chemical/L} = (\mu g \text{ chemical/g}_{OC}) \times 10^3 g_{OC}/Kg_{OC} \div K_{OC}$$

For chemicals with log K_{OW} greater than 5.5, corrections for DOC binding in the interstitial water will be necessary (see Gschwend and Wu 1985; Burkhard 2000). If the measured chemical in the interstitial water is substantially less (e.g., 2-3 fold lower or more), it suggests that the organic carbon in that sediment may not partition similarly to more typical organic carbon, and derivation of site-specific ESBs based on interstitial water may be warranted (U.S. EPA 2003b).

6.3 Summary

Based on the level of protection provided by WQC, the procedures described in this document indicate that benthic organisms should be comparably protected from adverse effects of endrin where endrin concentrations in sediment are below the ESB_{WQC} values of 5.4 µg endrin/g_{OC} for freshwater sediments and 0.99 µg endrin/g_{OC} for marine/estuarine sediments, except possibly where a locally important species is very sensitive or sediment organic carbon is $< 0.2\%$.

The ESBs do not consider the antagonistic, additive or synergistic effects of other sediment contaminants in combination with endrin or the potential for bioaccumulation and trophic transfer of endrin to aquatic life, wildlife or humans. Consistent with the recommendations of EPA's Science Advisory Board, publication of these documents does not imply the use of ESBs as stand-alone, pass-fail criteria for all applications; rather, exceedances of ESBs could trigger collection of additional assessment data.

Section 7

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Appendix A

Summary of Acute Values for Endrin used to Calculate the WQC FCV for Freshwater and Saltwater Species

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
FRESHWATER SPECIES									
Oligochaete worm, *Lumbriculus variegatus*	A	I	FT	M	>165.1	—	—	—	Poirier and Cox, 1991
Oligochaete worm, *Lumbriculus variegatus*	A	I	FT	M	>165.0	>165.0	>165.0	>165.0	Brooke, 1993
Cladoceran, *Simocephalus serrulatus*	X	W,E	S	U	26	—	—	—	Sanders and Cope, 1966; Mayer and Ellersieck, 1986
Cladoceran, *Simocephalus serrulatus*	X	W,E	S	U	45	34.20	34.20	34.20	Sanders and Cope, 1966; Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	L	W	S	U	4.2	—	—	—	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	L	W	S	U	74	—	—	—	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	L	W	S	U	41	—	—	—	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia magna*	L	W	FT	M	230	—	—	—	Thurston et al., 1985
Cladoceran, *Daphnia magna*	L	W	FT	M	88	142.3	—	—	Thurston et al., 1985
Cladoceran, *Daphnia pulex*	L	W	S	U	20	20	53.35	53.35	Mayer and Ellersieck, 1986
Ostracod, *Cypridopsis* sp.	A	I,E	S	U	1.8	1.8	1.8	1.8	Mayer and Ellersieck, 1986
Sowbug, *Asellus brevicaudus*	A	E	S	U	1.5	1.5	1.5	1.5	Sanders, 1972; Mayer and Ellersieck, 1986

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Scud, *Gammarus fasciatus*	A	E	S	U	4.3	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	X	E	S	U	1.3	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	X	E	FT	U	5.5	3.133	—	—	Sanders, 1972
Scud, *Gammarus lacustris*	A	E	S	U	3.0	3.0	3.066	3.066	Sanders, 1972; Mayer and Ellersieck, 1986
Glass shrimp, *Palaemonetes kadiakensis*	A	E	S	U	3.2	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Glass shrimp, *Palaemonetes kadiakensis*	X	E	FT	U	0.5	1.265	1.265	1.265	Sanders, 1972; Mayer and Ellersieck, 1986
Crayfish, *Orconectes immunis*	J	E	FT	M	>89	>89	—	—	Thurston et al., 1985
Crayfish, *Orconectes nais*	X	E	S	U	320	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Crayfish, *Orconectes nais*	J	E	S	U	3.2	3.2	3.2	16.88	Sanders, 1972; Mayer and Ellersieck, 1986
Mayfly, *Baetis* sp.	J	I	S	U	0.90	0.90	0.90	0.90	Mayer and Ellersieck, 1986
Mayfly, *Hexagenia bilineata*	J	I	S	U	62	62.99	62.99	62.99	Mayer and Ellersieck, 1986
Mayfly, *Hexagenia bilineata*	X	I	S	U	64	—	—	—	Sanders, 1972

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Stonefly, *Acroneuria* sp.	L	W,E	S	U	>0.18	>0.18	>0.18	>0.18	Mayer and Ellersieck, 1986
Stonefly, *Pteronarcella badia*	L	I,E	S	U	0.54	0.54	0.54	0.54	Sanders and Cope, 1968; Mayer and Ellersieck, 1986
Stonefly, *Pteronarcys californica*	A	I,E	S	U	0.25	0.25	0.25	0.25	Sanders and Cope, 1968; Mayer and Ellersieck, 1986
Stonefly, *Claassenia sabulosa*	J	W,E	S	U	0.76	—	—	—	Sanders and Cope, 1968
Stonefly, *Claassenia sabulosa*	J	W,E	S	U	0.76	0.2403	0.2403	0.2403	Mayer and Ellersieck, 1986
Caddis fly, *Brachycentrus americanus*	X	E	FT	M	0.34	0.34	0.34	0.34	Anderson and DeFoe, 1980
Damesfly, *Ischnura verticalus*	X	W,E	S	U	1.8	—	—	—	Sanders, 1972
Damesfly, *Ischnura verticalus*	J	W,E	S	U	2.1	—	—	—	Mayer and Ellersieck, 1986
Damesfly, *Ischnura verticalus*	J	W,E	S	U	2.4	2.086	2.086	2.086	Mayer and Ellersieck, 1986
Midge, *Tanytarsus dissimilis*	L	I	FT	M	0.83	0.83	0.83	0.83	Thurston et al., 1985
Diptera, *Tipula* sp.	J	I,E	S	U	12	12	12	12	Mayer and Ellersieck, 1986
Diptera, *Atherix variegata*	J	I,E	S	U	4.6	4.6	4.6	4.6	Mayer and Ellersieck, 1986
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	0.51	—	—	—	Katz, 1961

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	0.089	—	—	—	Mayer and Ellersieck, 1986
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	0.27	0.2306	—	—	Katz and Chadwick, 1961
Cutthroat trout, *Oncorhynchus clarki*	J	W	S	U	>1.0	>1.0	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	0.74	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	0.75	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	0.75	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	2.4	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	1.4	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	1.11	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	1.1	—	—	—	Macek et al., 1969
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	0.58	—	—	—	Katz, 1961
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	0.90	—	—	—	Katz and Chadwick, 1961
Rainbow trout, *Oncorhynchus mykiss*	J	W	FT	M	0.33	0.33	—	—	Thurston et al., 1985
Chinook salmon, *Oncorhynchus tshawytscha*	J	W	S	U	1.2	—	—	—	Katz, 1961

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L)				Reference
					Test	HMAV		Overall GMAV[h]	
						Species[f]	Genus[g]		
Chinook salmon, *Oncorhynchus tshawytscha*	J	W	S	U	0.92	1.051	>0.5318	>0.5318	Katz and Chadwick, 1961
Goldfish, *Carassius auratus*	J	W	S	U	2.1	—	—	—	Henderson et al., 1959
Goldfish, *Carassius auratus*	J	W	FT	U	0.44	—	—	—	Mayer and Ellersieck, 1986
Goldfish, *Carassius auratus*	J	W	FT	M	0.95	0.95	0.95	0.95	Thurston et al., 1985
Carp, *Cyprinus carpio*	J	W	FT	U	0.32	0.32	0.32	0.32	Mayer and Ellersieck, 1986
Fathead minnow, *Pimephales promelas*	J	W	S	U	1.1	—	—	—	Henderson et al., 1959
Fathead minnow, *Pimephales promelas*	J	W	S	U	1.4	—	—	—	Henderson et al., 1959
Fathead minnow, *Pimephales promelas*	L	W	S	U	0.7	—	—	—	Jarvinen et al., 1988
Fathead minnow, *Pimephales promelas*	J	W	S	U	1.8	—	—	—	Mayer and Ellersieck, 1986
Fathead minnow, *Pimephales promelas*	J	W	FT	U	0.24	—	—	—	Mayer and Ellersieck, 1986
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.50	—	—	—	Brungs and Bailey, 1966
Fathead minnow, *Pimephales promelas*	U	—	FT	M	0.49	—	—	—	Brungs and Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.40	—	—	—	Brungs and Bailey, 1966

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.45	—	—	—	Brungs and Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.64	0.4899	0.4899	0.4899	Thurston et al., 1985
Black bullhead, *Ictalurus melas*	J	W,E	S	U	1.13	—	—	—	Mayer and Ellersieck, 1986
Black bullhead, *Ictalurus melas*	J	W,E	FT	M	0.45	0.45	—	—	Anderson and DeFoe, 1980
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	0.32	—	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	1.9	—	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	0.8	—	—	—	McCorkle et al., 1977
Channel catfish, *Ictalurus punctatus*	J	W,E	FT	M	0.43	—	—	—	Thurston et al., 1985
Channel catfish, *Ictalurus punctatus*	J	W,E	FT	M	0.41	0.4199	0.4347	0.4347	Thurston et al., 1985
Flagfish, *Jordanella floridae*	J	W	FT	M	0.85	0.85	0.85	0.85	Hermanutz, 1978; Hermanutz et al., 1985
Mosquitofish, *Gambusia affinis*	J	W	S	U	1.1	—	—	—	Mayer and Ellersieck, 1986
Mosquitofish, *Gambusia affinis*	X	W	S	U	0.75	—	—	—	Katz and Chadwick, 1961

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L)				Reference
					Test	HMAV		Overall GMAV[h]	
						Species[f]	Genus[g]		
Mosquitofish, *Gambusia affinis*	J	W	FT	M	0.69	0.69	0.69	0.69	Thurston et al., 1985
Guppy, *Poecilia reticulata*	X	W	S	U	0.90	—	—	—	Katz and Chadwick, 1961
Guppy, *Poecilia reticulata*	X	W	S	U	1.6	1.200	1.200	1.200	Henderson et al., 1959
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.60	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	8.25	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	5.5	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	2.4	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	1.65	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.86	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.33	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.61	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.41	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.37	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.53	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.73	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.68	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.19	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.66	—	—	—	Henderson et al., 1959
Bluegill, *Lepomis macrochirus*	U	—	S	U	0.61	—	—	—	Sanders, 1972
Bluegill, *Lepomis macrochirus*	J	W	FT	M	0.19	—	—	—	Thurston et al., 1985
Bluegill, *Lepomis macrochirus*	J	W	FT	M	0.23	—	—	—	Thurston et al., 1985
Largemouth bass, *Micropterus dolomieu*	J	W	S	U	0.31	0.31	0.31	0.31	Mayer and Ellersieck, 1986
Yellow perch, *Perca flavescens*	J	W	FT	U	0.15	0.15	0.15	0.15	Mayer and Ellersieck, 1986
Tilapia, *Tilapia mossambica*	J	W	S	U	<5.6	<5.6	<5.6	<5.6	Mayer and Ellersieck, 1986
Bullfrog, *Rana catesbiana*	L	E	FT	M	2.5	2.5	—	—	Thurston et al., 1985
Southern leopard frog, *Rana sphenocephala*	E	W	FT	M	25	25	2.5(E) 25(W)	7.906	Hall and Swineford, 1980
Fowler's toad, *Bufo fowleri*	L	E	S	U	120	120	120	120	Mayer and Ellersieck, 1986

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Western chorus frog, *Psuedocris triseriata*	L	E	S	U	180	180	180	180	Mayer and Ellersieck, 1986
SALTWATER SPECIES									
Eastern oyster, *Crassostrea virginica*	E,L	W	S	U	790[i]	790	790	790	Davis and Hidu, 1969
Sand shrimp, *Crangon septemspinosa*	A	E	S	U	1.7	1.7	1.7	1.7	Eisler, 1969
Hermit crab, *Pagurus longicarpus*	A	E	S	U	12	12	12	12	Eisler, 1969
Korean shrimp, *Palaemon macrodactylus*	A	W,E	S	U	4.7	—	—	—	Schoettger, 1970
Korean shrimp, *Palaemon macrodactylus*	A	W,E	FT	U	0.3	1.187	1.187	1.187	Schoettger, 1970
Grass shrimp, *Palaemonetes pugio*	L	W	FT	M	1.2	—	—	—	Tyler-Schroeder, 1979
Grass shrimp, *Palaemonetes pugio*	J	W	FT	M	0.35	—	—	—	Tyler-Schroeder, 1979
Grass shrimp, *Palaemonetes pugio*	A	W,E	FT	M	0.69	—	—	—	Tyler-Schroeder, 1979
Grass shrimp, *Palaemonetes pugio*	A	W,E	FT	M	0.63	0.6536	—	—	Schimmel et al., 1975
Grass shrimp, *Palaemonetes vulgaris*	A	W,E	S	U	1.8	1.8	1.085	1.085	Eisler, 1969
Pink shrimp, *Penaeus duorarum*	A	I,E	FT	M	0.037	0.037	0.037	0.037	Schimmel et al., 1975
American eel, *Anguilla rostrata*	J	E	S	U	0.6	0.6	0.6	0.6	Eisler, 1969

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Chinook salmon, *Oncorhynchus tshawytscha*	J	W	FT	U	0.048	0.048	0.048	0.048	Schoettger, 1970
Sheepshead minnow, *Cyprinodon variegatus*	J	W,E	FT	M	0.37	—	—	—	Hansen et al., 1977
Sheepshead minnow, *Cyprinodon variegatus*	J	W,E	FT	M	0.34	—	—	—	Hansen et al., 1977
Sheepshead minnow, *Cyprinodon variegatus*	A	W,E	FT	M	0.36	—	—	—	Hansen et al., 1977
Sheepshead minnow, *Cyprinodon variegatus*	J	W,E	FT	M	0.38	0.3622	0.3622	0.3622	Schimmel et al., 1975
Mummichog, *Fundulus heteroclitus*	A	W,E	S	U	0.6	—	—	—	Eisler, 1970b
Mummichog, *Fundulus heteroclitus*	A	W,E	S	U	1.5	0.9487	—	—	Eisler, 1970b
Striped killifish, *Fundulus majalis*	J	W,E	S	U	0.3	0.3	0.5334	0.5334	Eisler, 1970b
Sailfin molly, *Poecilia latipinna*	A	W	FT	M	0.63	0.63	0.63	0.63	Schimmel et al., 1975
Atlantic silverside, *Menidia menidia*	J	W	S	U	0.05	0.05	0.05	0.05	Eisler, 1970b
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.65	—	—	—	Katz and Chadwick, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.50	—	—	—	Katz and Chadwick, 1961

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.20	—	—	—	Katz and Chadwick, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.57	—	—	—	Katz and Chadwick, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.57	—	—	—	Katz and Chadwick, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	0.44	—	—	—	Katz, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	0.50	1.070	1.070	1.070	Katz, 1961
Striped bass, *Morone saxatilis*	J	E	FT	U	0.094	0.094	0.094	0.094	Korn and Earnest, 1974
Shiner perch, *Cymatogaster aggregata*	J	W	S	U	0.8	—	—	—	Earnest and Benville, 1972
Shiner perch, *Cymatogaster aggregata*	J	W	FT	U	0.12	0.3098	0.3098	0.3098	Earnest and Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	S	U	0.6	—	—	—	Earnest and Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	FT	U	0.13	0.2793	0.2793	0.2793	Earnest and Benville, 1972
Bluehead, *Thalassoma bifasciatum*	A	W	S	U	0.1	0.1	0.1	0.1	Eisler, 1970b

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (μg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Striped mullet, *Mugil cephalus*	A	E	S	U	0.3	0.3	0.3	0.3	Eisler, 1970b
Northern puffer, *Sphaeroides maculatus*	A	W	S	U	3.1	3.1	3.2	3.1	Eisler, 1970b

[a]Life-stage: A = adult, J = juvenile, L = larvae, E = embryo, U = life-stage and habitat unknown, X = life-stage unknown but habitat known.
[b]Habitat: I = infauna, E = epibenthic, W = water column.
[c]Method: S = static, R = renewal, FT = flow-through.
[d]Concentration: U = unmeasured (nominal), M = chemical measured.
[e]Acute value: 96-hour LC50 or EC50, except for 48-hour EC50 for cladocera, barnacles, and bivalve molluscs (Stephan et al., 1985).
[f]HMAV species: Habitat Mean Acute Value - Species is the geometric mean of acute values by species by habitat (epibenthic, infaunal, and water column).
[g]HMAV genus: Geometric mean of HMAV for species within a genus.
[h]Overall GMAV: Geometric mean of acute values across species, habitats, and life-stages within the genus.
[i]Abnormal development of oyster larvae.

Appendix B

Summary of Data from Sediment-Spiking Experiments with Endrin (Data from these experiments were used to calculate K_{OC} values (Figure 2-2) and to compare mortalities of amphipods with interstitial water toxic units (Figure 4-1) and predicted sediment toxic units (Figure 4-2)).

Sediment Source, *Species tested*	Mortality (%)	Sediment Concentration (μg/g) Dry Weight	Sediment Concentration (μg/g) Organic Carbon	Interstitial Water Concentration[a] (μg/L)	TOC (%)	Log K_{OC}[b]	References
Soap Creek Pond	20	2.2	73	1.1	3.0	4.82	Nebeker et al.,
No. 7, OR	32	3.4	113	1.5	3.0	4.88	1989
Hyalella azteca	90	8.1	270	4.7	3.0	4.76	
	100	17.9	597	9.8	3.0	4.78	
	100	45.9	1,530	23.8	3.0	4.81	
1:1 Mixture Soap	9	1.1	18	0.5	6.1	4.56	Nebeker et al.,
Creek Pond And	44	4.9	80	1.7	6.1	4.67	1989
Mercer Lake, OR	95	17.7	290	6.8	6.1	4.63	
Hyalella azteca	100	31.7	520	10.6	6.1	4.69	
	100	56.4	924	24.5	6.1	4.58	
Mercer Lake, OR	5	1.1	10	0.3	11.2	4.59	Nebeker et al.,
Hyalella azteca	2	1.3	12	0.3	11.2	4.60	1989
	52	6.7	60	2.3	11.2	4.42	
	100	26.8	239	7.2	11.2	4.52	
	100	73.8	659	15.6	11.2	4.63	
Soap Creek	1.5	3.0	100	1.1	3.0	4.96	Schuytema et
Pond, OR	8.5	8.7	290	3.1	3.0	4.97	al., 1989
Hyalella azteca	100	19.6	653	6.1	3.0	5.03	
	100	40.4	1,350	13.9	3.0	4.99	
	100	62.1	2,070	22.2	3.0	4.97	
Mercer Lake, OR	10	2.0	18	0.4	11.0	4.65	Schuytema et
Hyalella azteca	5	5.3	48	1.0	11.0	4.68	al., 1989
	25	13.3	121	2.4	11.0	4.70	
	45	13.3	121	3.2	11.0	4.58	
	100	100	909	20.1	11.0	4.66	
Mercer Lake, OR	100	267	2,430	65.0	11.0	4.57	Schuytema et
Hyalella azteca	2.5	1.3	12	0.3	11.0	4.60	al., 1989
	12.5	1.3	12	0.2	11.0	4.60	
	10	8.0	73	0.8	11.0	4.96	
	100	20.0	182	3.9	11.0	4.67	
	100	66.7	606	10.8	11.0	4.75	
Lake Michigan	—	0.012[b]	17[b]	1.07	0.07	4.20	Stehly, 1992
Diporeia sp.	—	0.171[b]	31[b]	2.20	0.55	4.15	
	—	0.224[b]	13[b]	0.63	1.75	4.31	
						MEAN = 4.67	
						SE = 0.04	

[a]Interstitial water concentrations from Schuytema et al. (1989) are concentrations of "soluble" endrin in water overlying sediments. Sediments were refrigerated prior to testing.

[b]K_{OC} (L/kg) = sediment concentration ($\mu g/g_{OC}$) ÷ calculated interstitial water concentration (μg/L) × 10^3 g/kg.

Appendix C

Quality Assurance Summary for the ESB Document: Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Endrin

All data were obtained either from the WQC document for endrin (USEPA, 1980) or from a comprehensive literature search completed in 1995.

All data used in the example benchmark calculations were evaluated for acceptability using the procedures outlined in the Stephan et al. (1985): *Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses*. Data not meeting the criteria were rejected. All calculations were made using the procedures in Stephan et al. (1985). All calculations were checked by at least one other EPA scientist and then the document was distributed for public comment. All data and intermediate values are presented in tables in the document, and all original data were made available as part of the public comment process. Any errors of omission or calculation discovered during the public comment process were corrected and included in the revised document and can be found in *Comment Response Document for the Proposed Equilibrium Partitioning Sediment Guidelines for the Protection of Benthic Organisms*. Office of Water, Office of Science & Technology, (U.S. EPA, 2000).

Hard copies of all literature cited in this document reside at ORD/NHEERL Atlantic Ecology Division - Narragansett, Rhode Island.