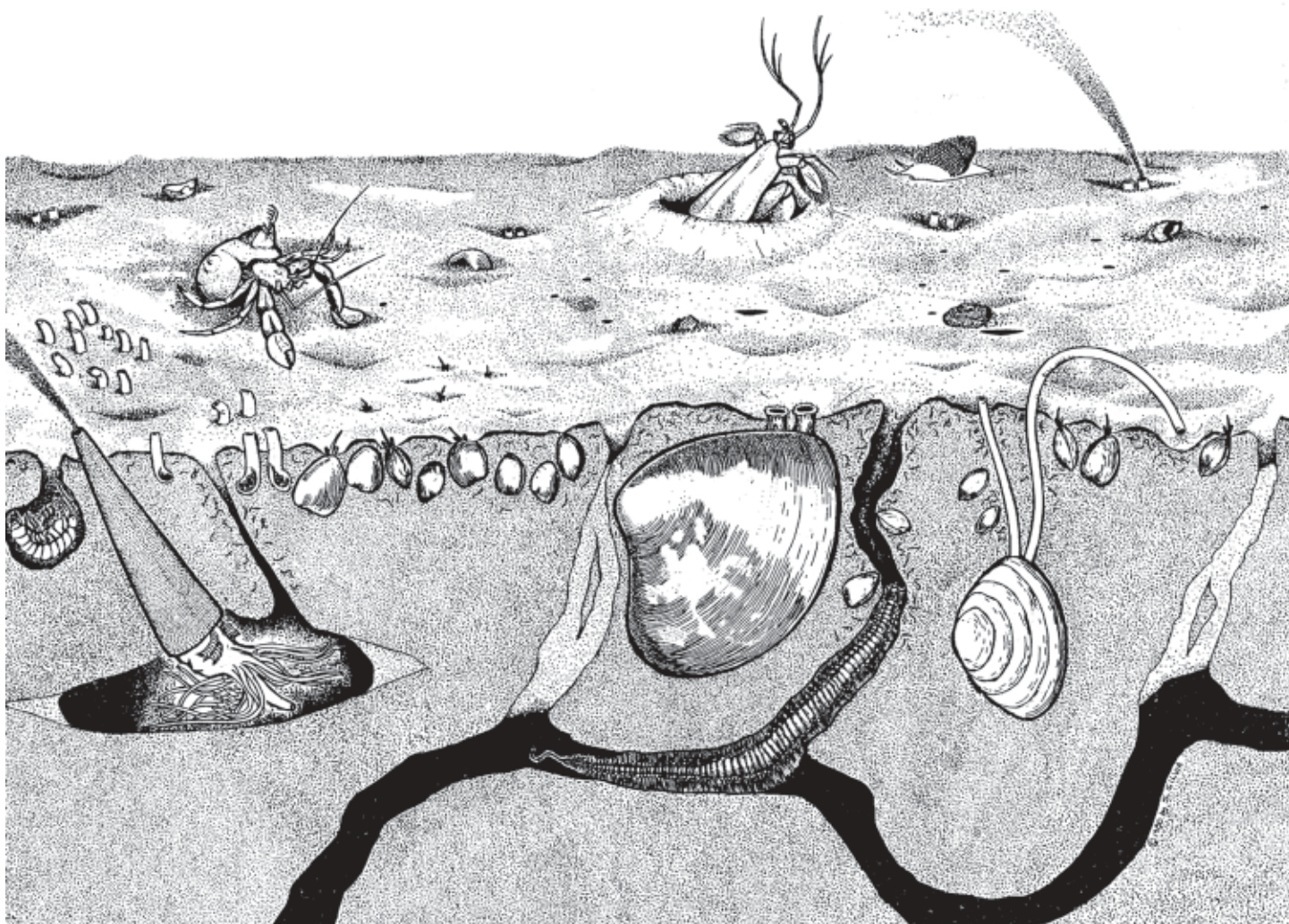# Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: Dieldrin

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

Walter J. Berry*
Robert M. Burgess**
National Health and Environmental Effects Research Laboratory
Atlantic Ecology Division
Narragansett, RI

David J. Hansen
HydroQual, Inc., Mahwah, NJ
Great Lakes Environmental Center, Traverse City, MI (formerly with U.S. EPA)

Dominic M. DiToro
Manhattan College, Riverdale, NY; HydroQual, Inc.,
Mahwah, NJ

Laurie D. DeRosa
HydroQual, Inc., Mahwah, NJ

Heidi E. Bell*
Mary C. Reiley
Office of Water, Washington, DC

Frank E. Stancil, Jr.
National Exposure Research Laboratory
Ecosystems Research Division
Athens, GA

Christopher S. Zarba
Office of Research and Development, Washington, DC

David R. Mount
Robert L. Spehar
National Health and Environmental Effects Research Laboratory
Mid-Continent Ecology Division
Duluth, MN

* Principle U.S. EPA Contacts
** Document Editor

U.S. Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Research Laboratory
Atlantic Ecology Division, Narragansett, RI
Mid-Continent Ecology Division, Duluth, MN

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 dieldrin. 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: Dieldrin. EPA-600-R-02-010. 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.

Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

Abstract

This equilibrium partitioning sediment benchmark (ESB) document describes procedures to derive concentrations of the insecticide dieldrin 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 dieldrin 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 dieldrin, this concentration is 12 µg dieldrin/g_{OC} for freshwater sediments and 28 µg/g_{OC} for saltwater sediments. Confidence limits of 5.4 to 27 µg/g_{OC} for freshwater sediments and 12 to 62 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 dieldrin or the potential for bioaccumulation and trophic transfer of dieldrin 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 dieldrin or the potential for bioaccumulation and trophic transfer of dieldrin 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.

This is contribution AED-02-047 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.

Contents

Contents

Tables

Figures

Acknowledgments

Executive Summary

This equilibrium partitioning sediment benchmark (ESB) document describes procedures to derive concentrations of the insecticide dieldrin 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 dieldrin 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 dieldrin, this concentration is 12 µg dieldrin/g_{OC} for freshwater sediments and 28 $\mu g/g_{OC}$ for saltwater sediments. Confidence limits of 5.4 to 27 $\mu g/g_{OC}$ for freshwater sediments and 12 to 62 $\mu g/g_{OC}$ for saltwater sediments were caluclated 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 dieldrin or the potential for bioaccumulation and trophic transfer of dieldrin 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 >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 dieldrin in this document, and for the nonionic organic insecticide endrin, 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
C_d	Freely-dissolved interstitial water chemical concentration
C_{IW}	Total interstitial water chemical concentration (includes freely-dissolved and DOC-complexed)
COE	U.S. Army Corps of Engineers
CFR	Code of Federal Regulations
CWA	Clean Water Act
DOC	Dissolved organic carbon
EC50	Chemical concentration estimated to cause adverse affects 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
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_{DOC}	Dissolved organic carbon partition coefficient
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
m_{DOC}	Measured DOC concentration
NAS	National Academy of Sciences
NERL	U.S. EPA, National Exposure Research Laboratory
NHEERL	U.S. EPA, National Health and Environmental Effects Research Laboratory
NOAA	National Oceanographic and Atmospheric Administration
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
SD	Standard deviation
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 sediment ESBs makes 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) 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, numerical 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 chemical concentrations in interstitial water.

3. The distribution of sensitivities to chemicals of benthic organisms 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 both 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) the 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 (μg/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 chemical concentrations in interstitial water may overestimate exposure.

Sediment benchmarks generated using the EqP approach (i.e., ESBs) are suitable for use in providing technical information to regulatory agencies because they are:

1. Numerical 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.

The ESB values presented herein are the concentrations of dieldrin 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 unprotective. 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 derivation of the ESBs for dieldrin. The data that support the EqP approach for deriving an ESB 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) and EPA (U.S. EPA, 1985, 2003a).

1.2   General Information: Dieldrin

Dieldrin is the common name of a persistent, nonsystemic organochlorine insecticide used for control of public health insect pests, termites, and locusts. It is formulated for use as an emulsifiable concentrate, as a wettable and dustable powder, or as a granular product. Another source of dieldrin in the environment other than from direct use of dieldrin stems from the quick transformation of aldrin, also an organochlorine pesticide, to dieldrin. Both dieldrin and aldrin usage peaked in the mid-1960s and declined until the early 1970s. All dieldrin products were canceled (including aldrin) in a PR notice, 71-4, dated March 18, 1971. See also Code of Federal Regulations (CFR) notice 37246, dated October 18, 1974.

Structurally, dieldrin is a cyclic hydrocarbon having a chlorine substituted methanobridge (Figure 1-1). It is similar to dieldrin, an endo-endo stereoisomer, and has similar physicochemical properties, except that it is more difficult to degrade in the environment (Wang, 1988). Dieldrin is a colorless crystalline solid at room temperature, with a melting point of about $176\,^\circ$C and specific gravity of 1.75 g/cc at $20\,^\circ$C. It has a vapor pressure of 0.4 mPa ($20\,^\circ$C) (Hartley and Kidd, 1987).

Dieldrin is considered to be toxic to aquatic organisms, bees, and mammals (Hartley and Kidd, 1987). The acute toxicity of dieldrin ranges from genus mean acute values (GMAVs) of 0.50 to 740 μg/L for freshwater organisms and 0.70 to 640 μg/L for saltwater organisms (Appendix A). Differences between dieldrin concentrations causing acute lethality and chronic toxicity in species acutely sensitive to this insecticide are small; acute–chronic ratios (ACRs) range from 1.189 to 11.39 for three species (see Table 3-2 in Section 3.3). Dieldrin bioconcentrates in aquatic animals from 400 to 68,000 times the concentration in water (U.S. EPA, 1980a). The WQC for dieldrin (U.S. EPA, 1980a) is

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 dieldrin 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	**1.75 g/cc (20°C)**
MELTING POINT	**176°C**
PHYSICAL FORM	**Colorless crystal**
VAPOR PRESSURE	**0.40 mPa (20°C)**

CAS NUMBER: 60-57-1
TSL NUMBER: IO 15750
COMMON NAME: Dieldrin (also dieldrine and ndieldrin)
TRADE NAME: Endrex (Shell); Hexadrin
CHEMICAL NAME: 1,2,3,4,10,10, hexachloro-1R, 4S, 4aS, 5R, 6R, 7S, 8SR, 8aR-octahydro-6, 7-epoxy-1, 4:5, 8-dimethanoaphthalene (IUPAC)

Figure 1-1. Chemical structure and physical-chemical properties of dieldrin (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 dieldrin in this document, the nonionic organic insecticide endrin (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 dieldrin. Section 2 reviews a variety of methods and data useful in deriving partition coefficients for dieldrin and includes the K_{OC} recommended for use in deriving the dieldrin ESB_{WQC}s. Section 3 reviews aquatic toxicity data contained in the dieldrin WQC document (U.S. EPA, 1980a) and new data that were used to derive the FCV used in this document to derive the ESB_{WQC}s concentrations. In addition, the comparative sensitivity of benthic and water column species is examined, and justification is provided for use of the FCV for dieldrin in the derivation of the ESB_{WQC}s. Section 4 reviews data on the toxicity of dieldrin in sediments, the need for organic carbon normalization of dieldrin 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 are used in Section 5 as the basis for the derivation of the ESB_{WQC}s for dieldrin and its uncertainty. The ESB_{WQC}s for dieldrin are then compared with three databases on dieldrin's environmental occurrence in sediments. Section 6 concludes with the sediment benchmarks for dieldrin 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 numerical 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 dieldrin 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 dieldrin 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 effect 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 the 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 dieldrin 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 in the sediment (f_{OC}). 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, the ESB_{WQC} term 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}), 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 dieldrin.

2.2   Determination of K_{OW} for Dieldrin

Several approaches have been used to determine K_{OW} for the derivation of an ESB_{WQC}, as discussed in the ESB_{WQC} Technical Basis Document. In an examination of the literature, primary references were found listing measured $\log_{10}K_{OW}$ values for dieldrin ranging from 4.09 to 6.20 and estimated $\log_{10}K_{OW}$ values ranging from 3.54 to 5.40 (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 5.37 for dieldrin from Karickhoff and Long (1995) is used to derive the ESB_{WQC} for dieldrin.

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 measurements of sediment dieldrin, sediment TOC, and calculated freely-dissolved concentrations of dieldrin in interstitial water were used to compute K_{OC}.

2.3.1 K_{OC} from Particle Suspension Studies

Laboratory studies to characterize adsorption 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

$$K_P = \frac{f_{OC}K_{OC}}{1 + mf_{OC}K_{OC}/\upsilon_X} \tag{2-4}$$

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

$$\log_{10}K_{OC} = 0.00028 + 0.983 \log_{10}K_{OW} \tag{2-5}$$

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 (Di Toro, 1985). The observed partition coefficient for dieldrin using adsorption data (Sharom et al., 1980) is highlighted on this plot. The observed $\log_{10}K_P$ of 1.68 reflects significant particle interaction effects. The observed partition coefficient is more than an order of

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

Method	$\text{Log}_{10}K_{OW}$	Reference
Measured	4.09	Ellington and Stancil, 1988
Measured	4.54	Brooke et al., 1986
Measured	4.65	De Kock and Lord, 1987
Measured	5.40	De Bruijn et al., 1989
Measured	6.20	Briggs, 1981
Estimated	3.54	Mabey et al., 1982
Estimated	5.40	Karickhoff et al., 1989

magnitude lower than the value expected in the absence of particle effects (i.e., $\log_{10}K_p = 3.32$ from the $f_{OC}K_{OC} = 2100$ L/kg). K_{OC} was computed from Equation 2-5.

Several sorption isotherm experiments with particle suspensions that provide an additional way to compute K_{OC} were found in a comprehensive literature search for partitioning information for dieldrin (Table 2-2). The K_{OC} values derived from these data are lower than K_{OC} values from laboratory measurements of K_{OW}. The lower K_{OC} can be explained from the particle interaction effects. Partitioning in a quiescent setting would result in less desorption and higher K_{OC}. These data are presented as examples of particle interaction if 100% reversibility is assumed in the absence of desorption studies and actual K_{OC} cannot be computed. In the absence of particle effects, K_{OC} is related to K_{OW} via

Equation 2-5. For $\log_{10}K_{OW} = 5.37$ (Karickhoff and Long, 1995), this expression results in an estimate of $\log_{10}K_{OC} = 5.28$.

2.3.2 K_{OC} from Sediment Toxicity Tests

Measurements of K_{OC} were available from sediment toxicity tests using dieldrin (Hoke and Ankley, 1992). These tests used a sediment having an average organic carbon content of 1.75% (Appendix B). Dieldrin concentrations were measured in sediments and in unfiltered interstitial waters, providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. Note that data from Hoke et al. (1995) were not used to calculate the partition coefficient because either interstitial water was not measured or free interstitial water could not be
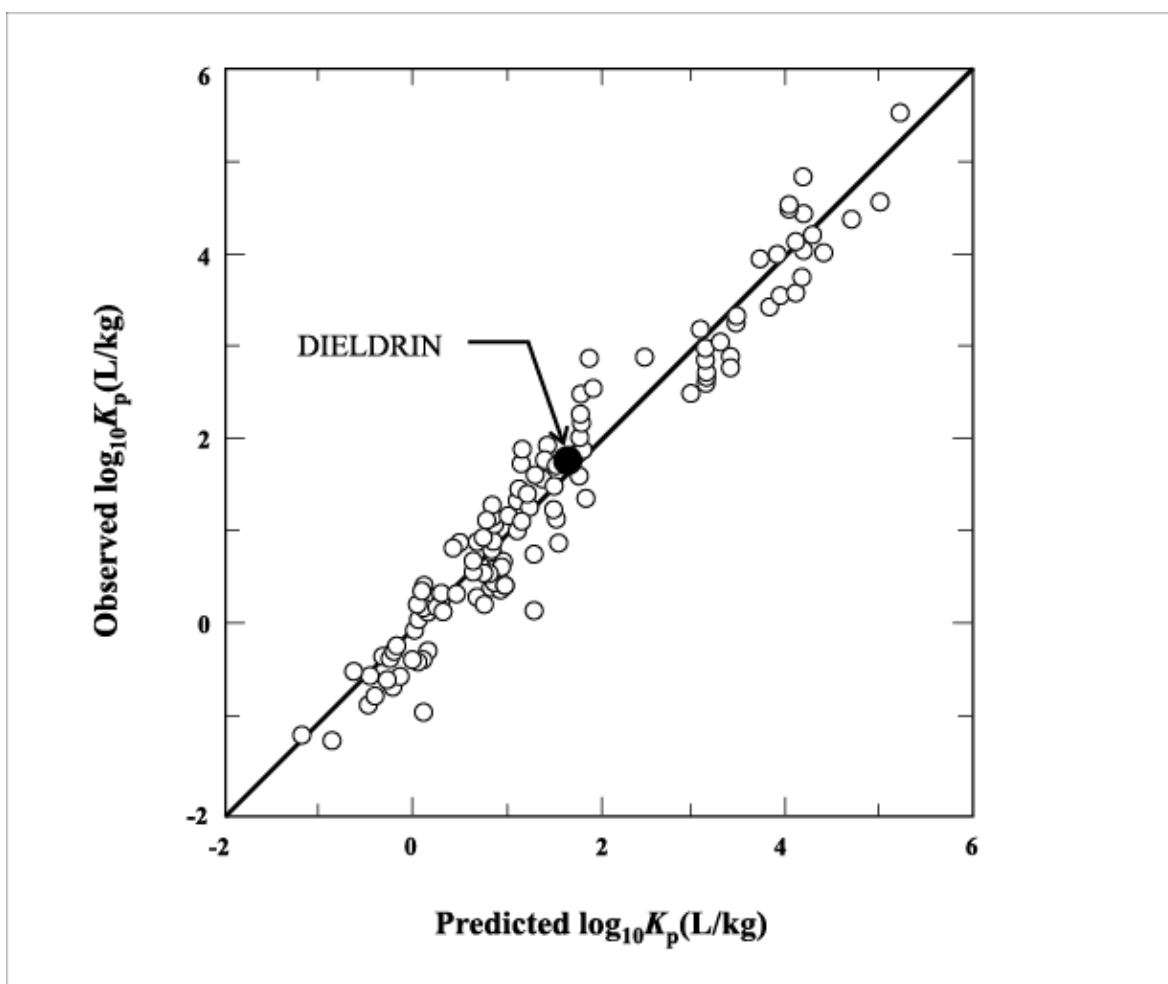
Figure 2-1. Observed versus predicted partition coefficients for nonionic organic chemicals, using Equation 2-4 (figure from Di Toro, 1985). Dieldrin datum is highlighted (Sharom et al., 1980).

Table 2-2. Summary of K_{OC} values for dieldrin derived from literature sorption isotherm data

Observed $\mathrm{Log}_{10}K_{OC}$ (SD)[a]	n	Solids (SD)[a] (g/L)	Reference
4.20 (0.14)	4	5.0	Eye, 1968
4.14 (0.15)	3	16.4 (4.6)	Betsill, 1990
4.10	1	100.0	Briggs, 1981

[a]SD = Standard deviation

correctly calculated. Since it is likely that organic carbon complexing in interstitial water is significant for dieldrin, organic carbon concentrations were also measured in interstitial water. Figure 2-2A is a plot of the organic carbon–normalized sorption isotherm for dieldrin, where the sediment dieldrin concentration (μg/ g_{OC}) is plotted versus the calculated free (dissolved) interstitial water concentration (μg/L). Using interstitial water dissolved organic carbon (DOC) concentrations, and assuming K_{DOC}, the dissolved organic carbon partition coefficient, is equal to K_{OC}, the calculated free interstitial water dieldrin concentration C_d (μg/L) presented in Figure 2-2 is given by

$$C_d = \frac{C_{IW}}{1 + m_{DOC}K_{DOC}} \tag{2-6}$$

where C_{IW} is the measured total interstitial water concentration and m_{DOC} is the measured DOC concentration (U.S. EPA, 2000a). The data used to make this plot are included in Appendix B. The line of unity slope corresponding to the $\mathrm{log}_{10}K_{OC} = 5.28$, derived from the dieldrin $\mathrm{log}_{10}K_{OW}$ of 5.37 from Karickhoff and Long (1995), is compared with the data. The data from the sediment toxicity tests fall on the line of unity slope for $\mathrm{log}_{10}K_{OC} = 5.28$ (Figure 2-2A).

A probability plot of the observed experimental $\mathrm{log}_{10}K_{OC}$ values is shown in Figure 2-2B. The $\mathrm{log}_{10}K_{OC}$ values were approximately normally distributed with a mean of $\mathrm{log}_{10}K_{OC} = 5.32$ and a standard error of the mean (SE) of 0.109. This value is in agreement with $\mathrm{log}_{10}K_{OC} = 5.28$, which was computed from the Karickhoff and Long (1995) dieldrin $\mathrm{log}_{10}K_{OW}$ of 5.37 (Equation 2-5).

2.4   Summary of Derivation of K_{OC} for Dieldrin

The K_{OC} selected to calculate the $\mathrm{ESB_{WQC}}$ for dieldrin was based on the regression of $\mathrm{log}_{10}K_{OC}$ to $\mathrm{log}_{10}K_{OW}$ (Equation 2-5) using the dieldrin $\mathrm{log}_{10}K_{OW}$ of 5.37 from Karickhoff and Long (1995). This approach, rather than the use of the K_{OC} from 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 $\mathrm{log}_{10}K_{OC} = 5.28$. This value is comparable to the $\mathrm{log}_{10}K_{OC}$ of 5.32 measured in the sediment toxicity tests.
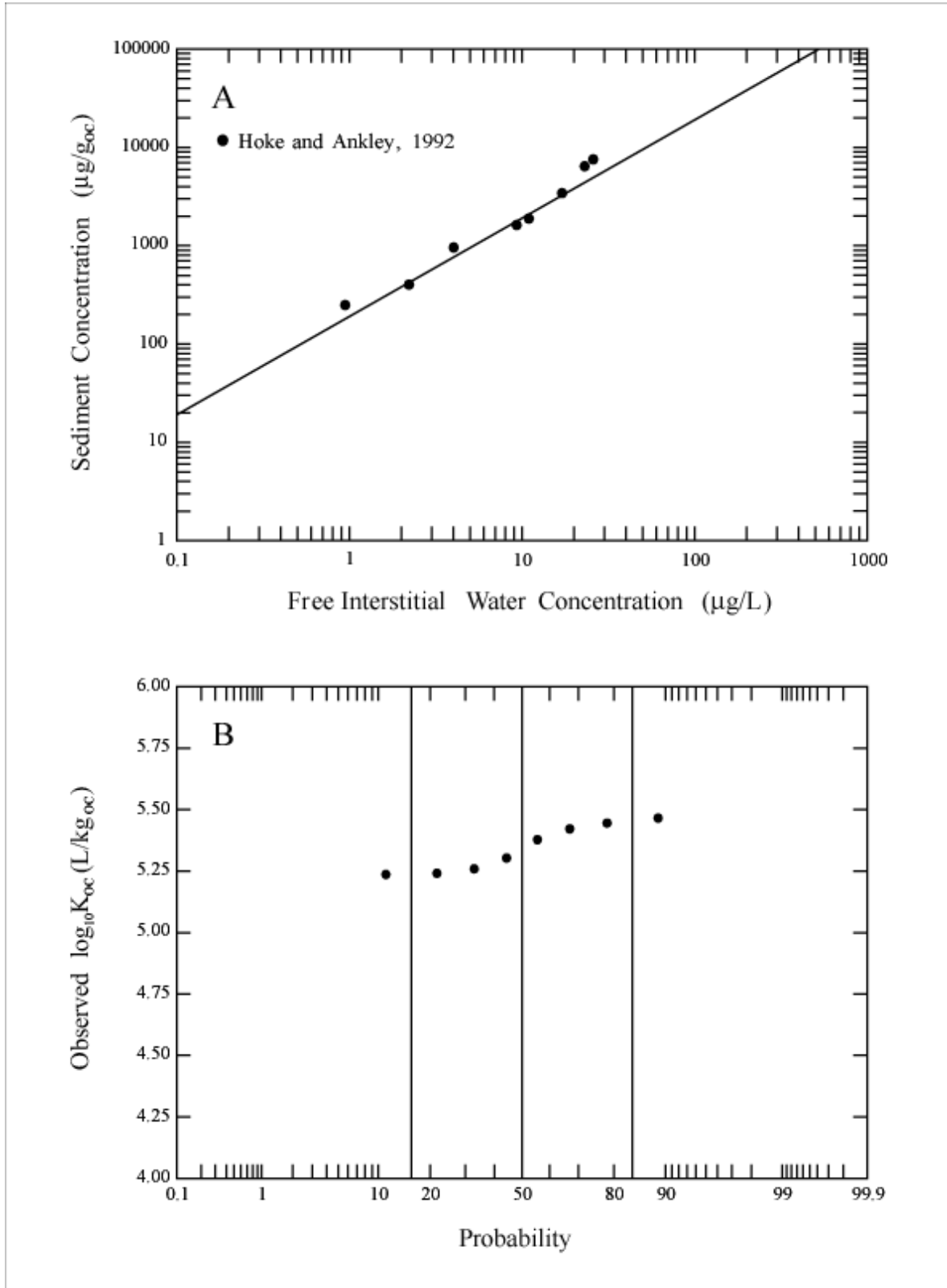
Figure 2-2. Organic carbon–normalized sorption isotherm for dieldrin (A) and probability plot of K_{OC} (B) from sediment toxicity tests (Hoke and Ankley, 1992). The solid line represents the relationship predicted with a $\log_{10}K_{OC}$ of 5.28.

Section 3

Toxicity of Dieldrin in Water Exposures

3.1 Derivation of Dieldrin WQC

The example used in Section 2 for the EqP method for derivation of the ESB_{WQC} for dieldrin uses the WQC FCV and K_{OC} to estimate the maximum concentrations of nonionic organic chemicals 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, 2000a). In this section, the FCV from the dieldrin WQC document (U.S. EPA, 1980a) 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 dieldrin ESB_{WQC}s.

3.2 Acute Toxicity in Water Exposures

A total of 116 standard acute toxicity tests with dieldrin have been conducted on 28 freshwater species from 21 genera (Figure 3-1; Appendix A). Of these tests, 38 were from 1 study with the guppy, *Poecilla reticulata* (Chadwick and Kiigemagi, 1968). Some of the values from this study have been omitted because they came from tests using water from generator columns that had not yet equilibrated. In some cases this may have led to toxicity related to unmeasured compounds, which the authors thought might have skewed the results. Similar logic was used to choose appropriate values in the WQC for dieldrin (U.S. EPA, 1980a). Overall GMAVs ranged from 0.5 to 740 µg/L. Stoneflies, fishes, isopods, damselflies, glass shrimp, and annelids were most sensitive; GMAVs for these taxa range from 0.5 to 21.8 µg/L. This database contained 18 tests on 15 benthic species from 13 genera (Figure 3-1; Appendix A).

Benthic organisms were both among the most sensitive and the most resistant freshwater species to dieldrin. GMAVs ranged from 0.5 to 740 µg/L. Of the epibenthic species tested, stoneflies, catfish, mayflies, isopods, and glass shrimp were most sensitive; GMAVs ranged from 0.5 to 20 µg/L. Infaunal species tested included only the oligochaete *Lumbriculus variegatus* (LC50=21.8 µg/L) and the stoneflies, *Pteronarcys californica* (LC50=0.5 µg/L) and *Pteronarcella badia* (LC50=0.5 µg/L). The LC50 represents the chemical concentrations estimated to be lethal to 50% of the test organisms within a specified time period.

A total of 29 acute tests have been conducted on 22 saltwater species from 20 genera (Figure 3-1; Appendix A). Overall GMAVs ranged from 0.70 to 640 µg/L. Sensitivities of saltwater organisms were similar to those of freshwater organisms. Fishes and crustaceans were the most sensitive. Within this database there were results from 20 tests on benthic life-stages of 15 species from 13 genera (Figure 3-2; Appendix A). Benthic organisms were among both the most sensitive and the most resistant saltwater genera to dieldrin. The most sensitive benthic species was the pink shrimp, *Peneaus duorarum*, with a measured flow-through 96-hour LC50 of 0.70 µg/L. The American eel, *Anquilla rostrata*, had a similar sensitivity to dieldrin, with a 96-hour LC50 of 0.9 µg/L. Other benthic species for which there were data appeared less sensitive, with GMAVs ranging from 4.5 to >100 µg/L.

3.3 Chronic Toxicity in Water Exposures

Chronic toxicity tests have been conducted with dieldrin using three freshwater fish and two saltwater invertebrates. The fish include rainbow trout, *Oncorhynchus mykiss*, the guppy, *P. reticulata*, and the fathead minnow, *Pimephales promelas*. The invertebrates include the mysid, *Americamysis bahia*, and the polychaete worm, *Ophryotrocha diadema* (Table 3-1). Both *O. mykiss* and *A.. bahia* have benthic life-stages.

Brooke (1993a) conducted an early life-stage test with *O. mykiss*. There were reductions of 35% in survival, 34% in weight, and 13% in length of the
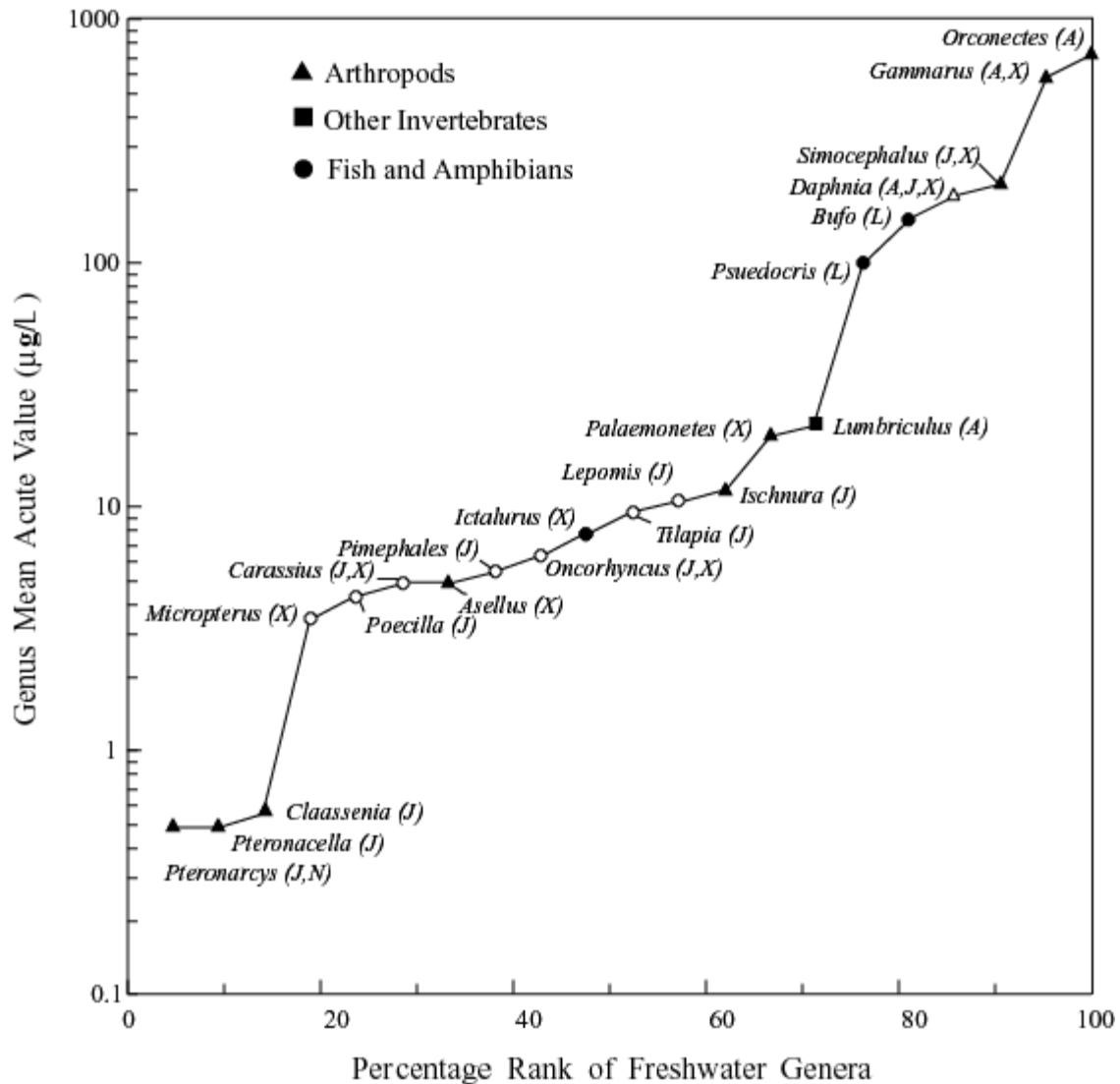
Figure 3-1. Genus mean acute values from water-only acute toxicity tests using freshwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. A=adult, J=juvenile, N=naiads, X=unspecified life-stage.

survivors in the 0.95 μg/L treatment relative to control fish. *O. mykiss* were not significantly affected at concentrations of 0.04 to 0.55 μg/L. The chronic value based on these results is 0.7228 μg/L. Combined with the 96-hour companion acute value of 8.23 μg/L (Brooke, 1993a), the ACR for this species is 11.39 μg/L (Table 3-2).

McCauley (1997) conducted an early life-stage test with the fathead minnow, *P. promelas*. There was a 91% reduction in survival in the 6.87 μg/L treatment relative to control fish. Fathead minnows were not significantly affected at concentrations of 0.38 to 3.02 μg/L. There were no effects on growth or reproduction recorded at any concentration tested. The chronic value based on these results is 4.555 μg/L. Two 96-hour LC50 tests were also conducted in the same dilution water as this test. One test was done with 30-day-old juveniles (LC50=4.45 μg/L), the other test was done with <24-hour-old larvae (LC50=6.59 μg/L). Because the LC50 values were from flow-through measured tests and were similar, the geometric mean of
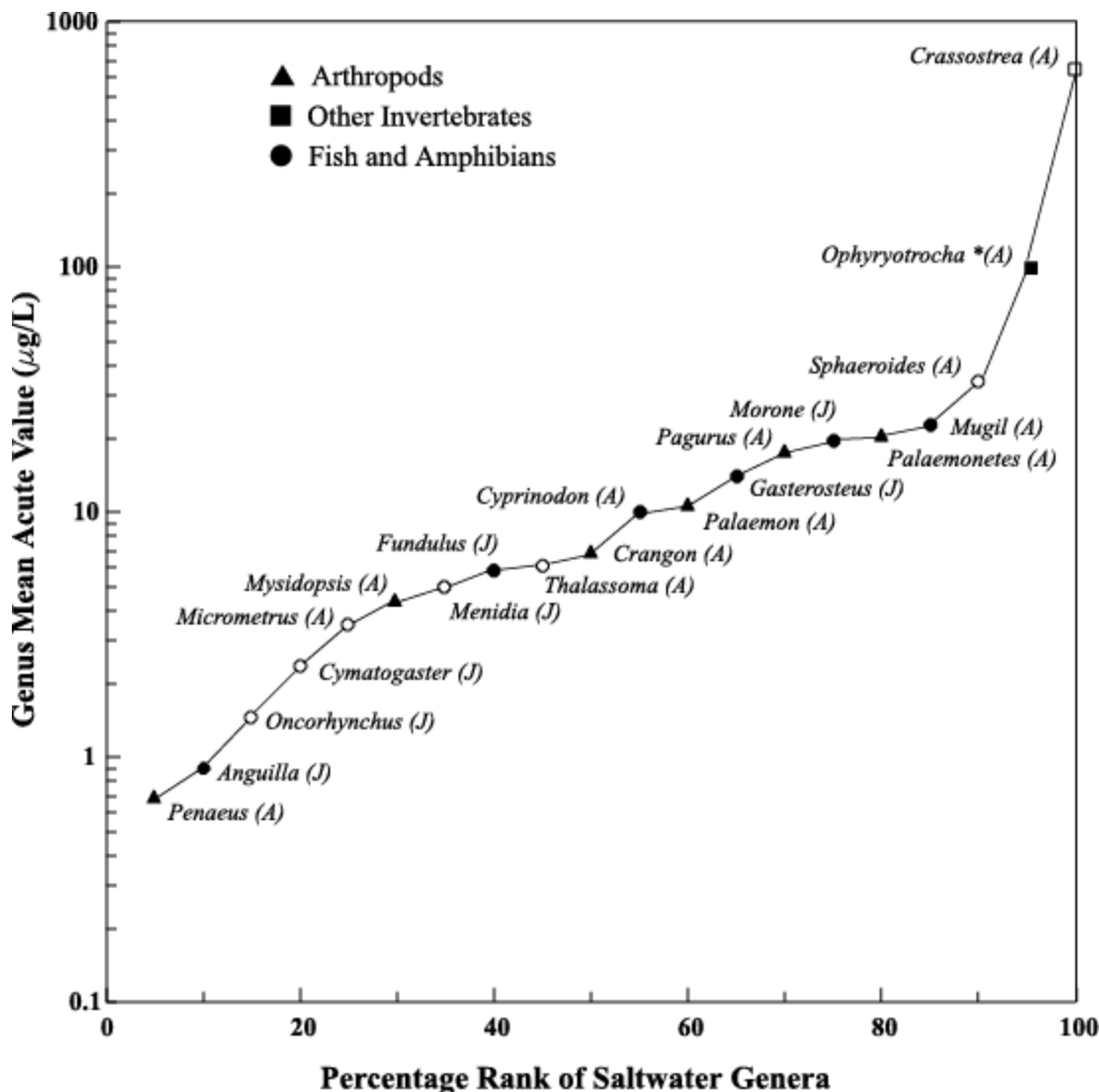
Figure 3-2. Genus mean acute values from water-only acute toxicity tests using saltwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. Asterisk indicates greater than values. A=adult, J=juvenile.

these two values (5.415 μg/L) was used in the calculation of the ACR, which is 1.189 μg/L for this species (Table 3-2).

Four freshwater chronic tests failed to meet the test requirement of a measured concentration for use in deriving WQC because there were no acceptable companion acute tests from the same dilution water. Therefore, the results of these tests were not used in the calculation of the final ACR (FACR). Although an ACR cannot be calculated from these data, the chronic

results are presented in Tables 3-1 and 3-2 to help establish the chronic effect levels of dieldrin for these species. One of these tests was an early life-stage test conducted with *O. mykiss* (Chadwick and Shumway, 1969). There were reductions of 97% in survival and 36% in growth of the survivors in the 0.39 μg/L treatment relative to control fish, and all fish died at 1.2 μg/L dieldrin. *Oncorhynchus mykiss* were not significantly affected at concentrations of 0.012 to 0.12 μg/L and no progeny were tested. The other freshwater chronic test that did not meet the "measured

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

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								
Rainbow trout, *Oncorhynchus mykiss*	ELS	W	100	0.012-0.12[d]	0.39, 1.2[d]	97-100% decrease in survival, 36% reduction in growth[e]	0.2163	Chadwick and Shumway, 1969
Rainbow trout, *Oncorhynchus mykiss*	ELS	W	28	0.04-0.55	0.95	35% decrease in survival, 13% reduction in length, 34% in weight	0.7228	Brooke, 1993a
Guppy, *Poecilia reticulata*	LC	W	195	0.05, 0.2, 1.0[d]	—	—	>1.0	Roelofs, 1971
Guppy, *Poecilia reticulata*	LC	W	195	0.2, 1.0, 2.5[d]	—	—	>2.5	Roelofs, 1971
Guppy, *Poecilia reticulata*	LC	W	195	0.2, 2.5[d]	1.0[d]	42% reduction in brood size	>2.5	Roelofs, 1971
Fathead minnow, *Pimephales promelas*	ELS	W	30	0.38-3.02	6.87	91% decrease in survival	4.555	McCauley, 1997
Saltwater Species								
Mysid, *Americamysis bahia*	LC	E (J,A)	28	0.10, 0.49	0.22, 1.1, 1.6	24-58% decrease in survival	0.7342	EPA, 1980b
Polychaete worm, *Ophryotrocha diadema*	LC	I (L)	47	0.1[f]	0.3-13	34% decrease in survival, 37-99% reduction in reproduction, 16-71% decrease in progeny survival	0.1732	Hooftman and Vink, 1980
Polychaete worm, *Ophryotrocha diadema*	PLC	I (A)	37	1.2	2.6-72	63% decrease in survival, 57-100% reduction in reproduction 39-100% decrease in progeny survival	1.766	Hooftman and Vink, 1980

[a]Test: LC = life-cycle, PLC = partial life-cycle, ELS = early life-stage.
[b]Habitat: I = infaunal, E = epibenthic, W = water column. Life-stage: E = embryo, L = larval, J = juvenile, A = adult.
[c]NOECs = No observed effect concentration(s); OECs = Observed effect concentration(s).
[d]Nominal, not measured.
[e]Estimated from graph.
[f]Nominal (less than limit of analytical detection); all other values listed are measured values (there was good agreement between nominal and measured).

3-4

Table 3-2.   **Summary of freshwater and saltwater acute and chronic values, acute–chronic ratios, and derivation of the final acute values, final acute–chronic ratios, and final chronic values for dieldrin**

Common Name, *Scientific Name*	Acute Value (96-hour) (μg/L)	Chronic Value (μg/L)	Acute-Chronic Ratio (ACR)	Species Mean Acute-Chronic Ratio (SMACR)
Freshwater Species				
Rainbow trout, *Oncorhynchus mykiss*		0.2163[a]		
Rainbow trout, *Oncorhynchus mykiss*	8.23	0.7228	11.39	11.39
Guppy, *Poecilia reticulata*		>1.0[a]		
Guppy, *Poecilia reticulata*		>2.5[a]		
Guppy, *Poecilia reticulata*		0.447[a]		
Fathead minnow, *Pimephales promelas*	5.415[b]	4.555	1.189	1.189
Saltwater Species				
Mysid, *Americamysis bahia*	4.5	0.7342	6.129	6.129
Polychaete worm, *Ophryotrocha diadema*	>100	0.1732	>577.4[c]	>577.4
Polychaete worm, *Ophryotrocha diadema*	>100	1.766	>56.63[c]	

[a]Not used in calculation of SMACR or FACR because acute value from matching dilution water is not available.
[b]Acute value geometric mean of test with 30-day-old juveniles and test with <24-hour-old fish in the same dilution water (see text).
[c]Not used in calculation of SMACR or FACR because ACRs are greater than values. Also because the range of ACRs, if these are included, is greater than a factor of 10.0, this species is much less acutely sensitive than the other species with available ACRs, and the FAV derived with the other three ACRs is protective of this species (see text).

Freshwater	Saltwater
Final acute value = 0.2874 μg/L	Final acute value = 0.6409 μg/L
Final acute-chronic ratio = 4.362	Final acute-chronic ratio = 4.362
Final chronic value = 0.06589 μg/L	Final chronic value = 0.1469 μg/L

concentrations" criteria was a three-generation study using the guppy, *P. reticulata* (Roelofs, 1971). Only data from three tests with the first-generation fish were included in Tables 3-1 and 3-2 because the test organisms in the second- and third-generation tests received some exposure prior to testing. There was no effect on *P. reticulata* survival at any dieldrin concentration in the first test (from 0.05 to 1.0 μg/L) or in the second test (from 0.2 to 2.5 μg/L). In the third test, mean brood size was reduced by 42% at 1.0 μg/L. The 32% reduction in growth at 2.5 μg/L was not

statistically significant. Because there were no statistically significant differences from controls at the highest concentration, the chronic value from this test is considered to be >2.5 μg/L.

Saltwater *A. bahia* exposed to dieldrin in a life-cycle test (U.S. EPA, 1987b) were affected at concentrations similar to those affecting the two freshwater fish mentioned above. Survival of *A. bahia* exposed to 0.22, 1.1, and 1.6 μg/L was reduced by 24%, 35%, and 58%, respectively, relative to control

A. bahia. There were no significant effects at 0.49 μg/L. No effects were observed on reproduction at any concentration tested, and progeny response was not recorded. Based on these results, the chronic value for *A. bahia* is 0.7342 μg/L. Combined with the 96-hour companion acute value of 4.5 μg/L (U.S. EPA, 1987b), the ACR for this species is 6.129 μg/L.

Two chronic tests were performed with saltwater organisms that could not be used in the calculation of the FACR because definitive companion acute values could not be calculated. One life-cycle test and one partial life-cycle test were conducted with the marine polychaete worm, *O. diadema* (Hooftman and Vink, 1980) (see Tables 3-1 and 3-2). The nominal no observed effect concentration (NOEC) was 0.1 μg/L (below the limit of analytical detection) for the life-cycle test initiated with larvae and 1.2 μg/L (based on measured concentrations) for the partial life-cycle test initiated with adults. For the life-cycle test with larvae, there was a 37% to 99% decrease in reproductive potential (combined effect on number of egg masses and embryo survival), relative to carrier control worms at 0.3 to 13 μg/L dieldrin. Progeny survival was reduced by 35%, 16%, 61%, and 71% at dieldrin concentrations of 0.3, 1.5, 3.1, and 13 μg/L, respectively. At 13 μg/L dieldrin, larval survival was reduced to 34% relative to the controls. The chronic value for this test was 0.1732 μg/L. In the *O. diadema* partial life-cycle test, reproductive potential was reduced by 57%, 92%, 97%, and 100% relative to the carrier control in concentrations of 2.6, 8.0, 23, and 72 μg/L, respectively. Of adults in 72 μg/L, 63% died. Reductions in egg survival were 39%, 70%, 62%, and 100% relative to controls in concentrations of 2.6, 8.0, 23, and 72 μg/L, respectively. The chronic value for this test was 1.766 μg/L, over an order of magnitude higher than that from the full life-cycle test. The chronic sensitivity of this species appeared similar to that of the other species tested chronically, but acute sensitivity was low: 96-hour LC50 >100 μg/L for adults and larvae. The FCV calculated using the ACRs available from other species is protective of this species.

The final acute value (FAV) derived from the overall GMAVs (Stephan et al., 1985) for freshwater organisms was 0.2874 μg/L (Table 3-2). The FAV derived from the overall GMAVs (Stephan et al., 1985) for saltwater organisms was 0.6409 μg/L (Table 3-2), less than the acute value for the economically important shrimp, *P. duorarum*. The available ACRs for three species were 1.189 for *P. promelas*, 6.129 for *A.*

bahia, and 11.39 for *O. mykiss*. The FACR, the geometric mean of these three values, was 4.362. The FCVs (Table 3-2) for calculating the ESB$_{WQC}$s for dieldrin were calculated by dividing both the freshwater and saltwater FAV by the FACR. The FCV for freshwater organisms of 0.06589 μg/L was the quotient of the FAV of 0.2874 μg/L and the FACR of 4.362. Similarly, the FCV for saltwater organisms of 0.1469 μg/L was the quotient of the FAV of 0.6409 μg/L and the FACR of 4.362.

3.4 Applicability of the WQC as the Effects Concentration for Derivation of Dieldrin ESB$_{WQC}$s

Use of the FCV as the effects concentration for calculation of the ESB$_{WQC}$ assumes that benthic (infaunal and epibenthic) species, as a group, have sensitivities similar to all benthic and water column species tested to derive the WQC concentration. 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 dieldrin-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 dieldrin used in this comparison.

For dieldrin, benthic species account for 13 out of 21 genera tested in freshwater and 13 of 20 genera tested in saltwater (Figures 3-1, 3-2, Appendix A). An initial test of the difference between the freshwater and saltwater FAVs for all species (water column and benthic) exposed to dieldrin 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 were 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 dieldrin, the test statistic occurred at the 16th percentile of the generated FAV differences. Because the probability was less than 95%, the hypothesis of no significant difference in sensitivity for freshwater and saltwater species was accepted (Table 3-3). Note that in both the freshwater versus saltwater comparison and benthic versus WQC comparison, greater than (>) values for GMAVs (see Appendix A) were omitted from the AR analysis. This resulted in one dieldrin saltwater benthic organism being omitted.

Because freshwater and saltwater species showed similar sensitivity, a test of difference in sensitivity was performed for benthic and all (benthic and water column species combined, hereafter referred to as "WQC") organisms combining freshwater and saltwater species, using the AR method. For this purpose, each life-cycle of each test organism was assigned a habitat (Appendix A) using the criteria observed by EPA (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 in the first test 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 a "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 Figure 3-3B. The test statistic for this analysis occurred at the 68th percentile, and the hypothesis of no difference in sensitivity was accepted (Table 3-3). This analysis suggests that the FCV for dieldrin 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 dieldrin 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 (21)	Salt (19)	−0.334	16
Benthic vs. Water Column = Benthic (WQC)	Benthic (26)	WQC (40)	0.052	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 one dieldrin saltwater benthic organism 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.
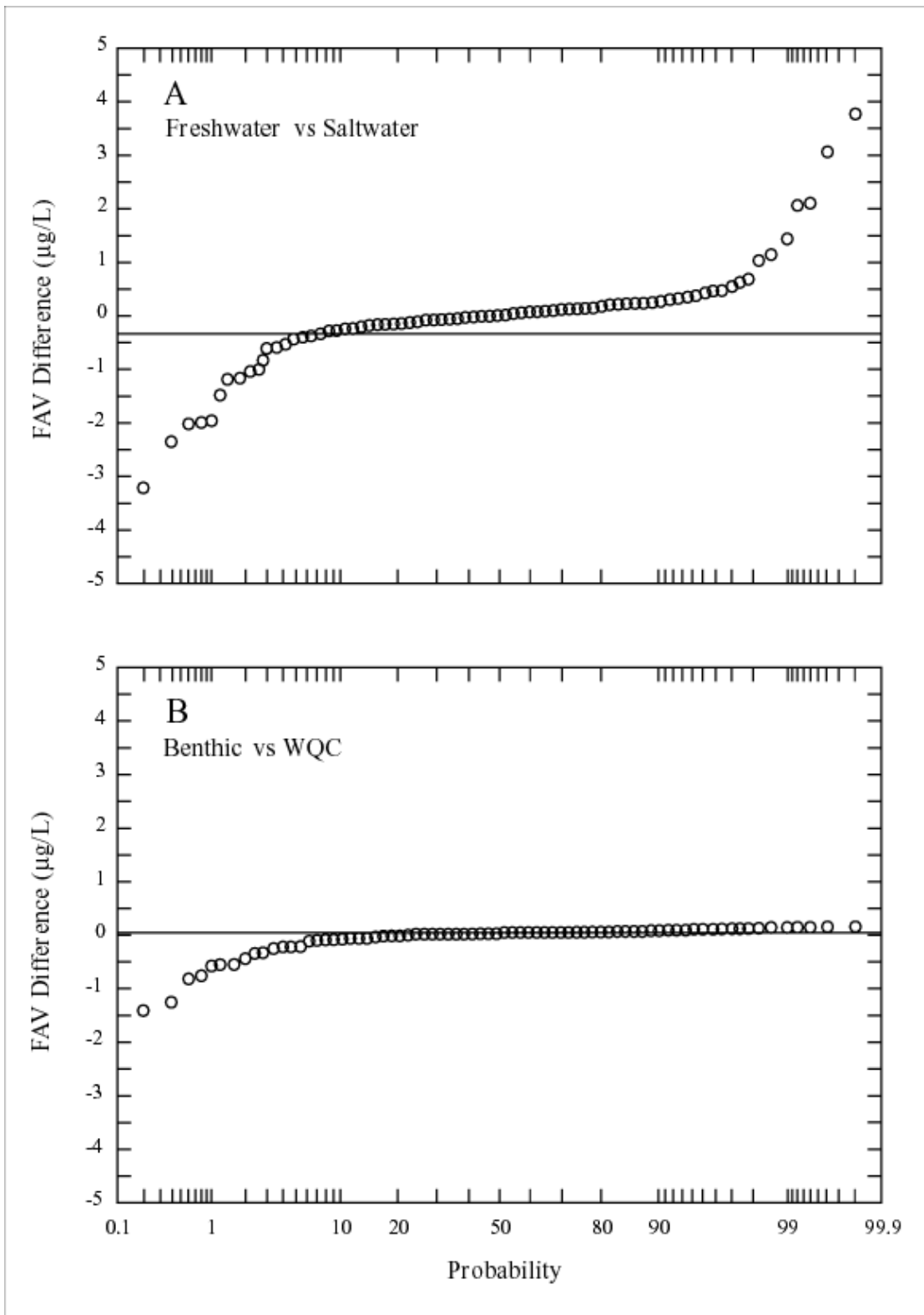
Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater versus saltwater (A) and benthic versus WQC (B) data. The solid lines in the figure correspond to the FAV differences measured for dieldrin.

Section 4

Actual and Predicted Toxicity of Dieldrin in Sediment Exposures

4.1 Toxicity of Dieldrin in Sediments

The toxicity of dieldrin-spiked clean sediments was tested with two freshwater species (an amphipod and a midge) and two saltwater species (a polychaete and the sand shrimp) (Table 4-1). Therefore, generalizations of dieldrin's toxicity across species or sediments are limited. The endpoint reported in these studies was mortality (with the addition of dry weight in the midge tests). Details about exposure methodology are provided because sediment testing methodologies have not been standardized in the way that water-only toxicity test methodologies have. Data were available from many experiments using both field and laboratory sediments contaminated with mixtures of dieldrin and other compounds. Data from these studies were not included here because it was not possible to determine the contribution of dieldrin to the observed toxicity.

The effects of sediment from three freshwater sites in Minnesota spiked with dieldrin on the freshwater amphipod, *H. azteca,* were studied by Hoke et al. (1995). The TOC concentrations in the three sediments were 1.7%, 2.9%, and 8.7%, respectively. The sediments were rolled in dieldrin-coated jars at 4°C for 23 days. Mortality of *H. azteca* in these flow-through tests was related to sediment exposure because dieldrin concentrations in overlying water were generally below detection limits. Given the "nonstandard" dose response in many of the tests with *H. azteca*, the LC50 values from these tests need to be examined carefully. In several of these tests, toxicity increased with concentration up to an intermediate concentration and then decreased with further increasing concentration. It may be that the amphipods were avoiding the sediment in the higher concentrations by coming out of the sediment, thereby limiting their exposure (R. Hoke, E.I. DuPont deNemours and Co., Haskell Laboratory, Newark, DE, personal communication). No dose-response relationship was observed in the results from the definitive test with one of the sediments (Airport Pond) or in the results from further testing with this sediment using *H. azteca* (Hoke et al., 1995). In at least one of the Airport Pond

sediment repeat experiments, mortality seemed to be increasing at a concentration similar to that causing 50% mortality in the range-finder test, and then dropped off. For this reason, only the Airport Pond data from the range-finder test with this sediment are used in the analysis of the toxicity data (Sections 4.1, 4.2, 4.3) and in Figures 4-1 and 4-2. The 10-day LC50 values increased with increasing TOC when dieldrin concentration was expressed on a dry weight basis, but increased only slightly with increasing organic carbon when dieldrin concentration was expressed on an organic carbon basis (Table 4-1). Hoke et al. (1995) calculated organic carbon–normalized concentrations based on TOC measured in individual treatments. This leads to the apparent discrepancy between the experiment mean TOC values and the organic carbon–normalized concentrations reported in Tables 4-1 and 4-2. LC50 values normalized to dry weight differed by a factor of 19.4 (22.8 to 441.8 μg/g) over a fivefold range of TOC. In contrast, the organic carbon–normalized LC50 values differed by a factor of 3.2 (1,322 to 4,272 μg/g$_{OC}$).

The effects of dieldrin-spiked sediments on the fresh water midge, *C. tentans*, were also reported by Hoke et al. (1995). The TOC contents in the two sediments were 1.5% and 2.0%. The sediments were rolled in dieldrin-coated jars at 4°C for 30 days, stored at 4°C for 60 days, and then rolled at 4°C for an additional 30 days. LC50 values normalized to dry weight differed by a factor of 3.0 (0.5 to 1.5 μg/g dry weight). LC50 values normalized to organic carbon differed by a factor of 2.7 (35.1 to 95.3 μg/g$_{OC}$). It is not surprising that organic carbon normalization had little effect, given the small range of TOC (1.5% to 2.0%).

The only saltwater experiments that tested dieldrin-spiked sediments were conducted by McLeese et al. (1982) and McLeese and Metcalfe (1980). These began with clean sediments that were added to dieldrin-coated beakers just before the addition of test organisms. This is a marked contrast with tests using

freshwater sediments spiked with dieldrin days or weeks prior to test initiation. As a result, the dieldrin concentrations in the sediment and overlying water varied greatly over the course of these saltwater experiments, and exposure conditions are uncertain. In addition, transfer of test organisms to freshly prepared beakers every 48 hours further complicates interpretation of results of McLeese et al. (1982), because exposure conditions changed several times during the course of the test. McLeese et al. (1982) tested the effects of dieldrin on the polychaete worm,

Nereis virens, in sediment with 2% TOC (17% sand and 83% silt and clay) in 12-day toxicity tests. No worms died in 13 μg/g dry weight sediment, the highest concentration tested. McLeese and Metcalfe (1980) tested the effects of dieldrin in sand with a TOC content of 0.28% on the sand shrimp, *Crangon septemspinosa*. The 4-day LC50 value was 0.0041 μg/g dry weight sediment (1.46 μg/g$_{OC}$). Concentrations of dieldrin in water overlying the sediment were 10 times the LC50 in water. The authors concluded that sediment-associated dieldrin contributed little to the

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

Common Name, *Scientific Name*	Sediment Source	TOC (%)	Method,[a] Duration (days)	Response	Sediment Dieldrin LC50 Dry wt (μg/g)	OC (μg/g)	Interstitial Water LC50 (μg/L)	Reference
Freshwater Species								
Amphipod, *Hyalella azteca*	Airport Pond, MN	1.7[b]	FT, M/10	LC50	22.8	1,332[c]	54.3	Hoke et al., 1995
Amphipod, *Hyalella azteca*	West Bearskin Lake, MN	2.9[b]	FT, M/10	LC50	43.4	1,322[c]	236	Hoke et al., 1995
Amphipod, *Hyalella azteca*	Pequaywan Lake, MN	8.7[b]	FT, M/10	LC50	441.8	4,272[c]	492	Hoke et al., 1995
Midge, *Chironomus tentans*	Airport Pond, MN	2.0[b]	FT, M/10	LC50	1.5	95.3[c]	0.5[d]	Hoke et al., 1995
Midge, *Chironomus tentans*	Airport Pond, MN	1.5[b]	FT, M/10	LC50	0.5	35.1[c]	0.2[d]	Hoke et al., 1995
Saltwater Species								
Polychaete worm, *Nereis virens*	17% sand, 83% silt and clay[e]	2.0	R, M/12	LC50	>13	>650	—	McLeese et al., 1982
Sand shrimp, *Crangon septemspinosa*	Sand, wet-sieved between 1-2 mm sieves[e]	0.28	R, M/4	LC50	0.0041	1.46	—	McLeese and Metcalfe, 1980

[a]FT = flow-through; M = measured; R = renewed.
[b]Mean reported TOC concentration.
[c]Calculated using individually measured TOC concentrations.
[d]Interstitial water concentrations estimated from f_{OC}, K_{OC}, and measured sediment concentrations.
[e]Clean sediment placed in dieldrin-coated beakers at beginning of exposure.

Figure 4-1. Percent mortalities 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 dieldrin (Hoke et al., 1995) or kepone (Adams et al., 1985) relative to interstitial water units.

toxicity observed.

The need for organic carbon normalization of the concentration of nonionic organic chemicals in sediments is presented in the ESB Technical Basis Document. For dieldrin, this need is supported by the dieldrin-spiked toxicity tests described above, particularly the experiments with *H. azteca* by Hoke et al. (1995). Although it is important to demonstrate that organic carbon normalization is necessary if benchmarks are to be developed 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 dieldrin and other nonionic organic chemicals on an organic carbon basis for a range of sediments. Evidence supporting this prediction for dieldrin and other nonionic organic chemicals is contained in the following sections.

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 content (U.S. EPA, 2003a). Measured or estimated interstitial water values were available from studies with two species (Table 4-2). Data from tests with water column species were not considered in this analysis. Hoke et al. (1995) found that 10-day LC50 values for *H. azteca* based on measured interstitial water concentrations differed by a factor of 9.1 (54.3 to 491.6 μg/L) for three sediments containing from 1.7% to 8.7% TOC. Therefore, interstitial water–normalized LC50 values provided an improvement over LC50 values for dieldrin expressed on a dry weight basis which varied by a factor of 19.4

(22.8 to 441.8 μg/g) (Table 4-1). The authors proposed partitioning to DOC to explain the small disparity between LC50 values based on interstitial water dieldrin concentrations (Hoke et al., 1995). They found that the 10-day LC50 values for *C. tentans* based on predicted interstitial water concentrations (the sediment concentration multiplied by the K_{OC}; used because measured concentrations were not available) differed by a factor of 2.8 (0.18 to 0.50). This variability was slightly less than that shown when dry weight was used (factor of 3.0), but similar to that shown when organic carbon normalization was used (factor of 2.7).

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 was made utilizing results from toxicity tests with sediments spiked with a variety of nonionic compounds, including acenaphthene and phenanthrene (Swartz, 1991), dieldrin (Hoke et al., 1995), 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 data included in the following analyses were from tests conducted at EPA laboratories or from tests that utilized designs at least as rigorous as those conducted at EPA laboratories. Tests with acenaphthene and phenanthrene used two saltwater amphipods (*Leptocheirus plumulosus* and *Eohaustorius estuaris*) and saltwater sediments. Tests with fluoranthene used a saltwater amphipod (*Rhepoxynius abronius*) and saltwater sediments. Freshwater sediments spiked with dieldrin and endrin were tested using the amphipod *H. azteca*, and kepone-spiked sediments and dieldrin-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 chemical versus interstitial water TUs (IWTUs) for all sediments tested with the following caveat for dieldrin. Only the *C. tentans* Airport Pond data are used for dieldrin, in part due to difficulties with the *H. azteca* mortality results, as previously discussed (Figure 4-1).

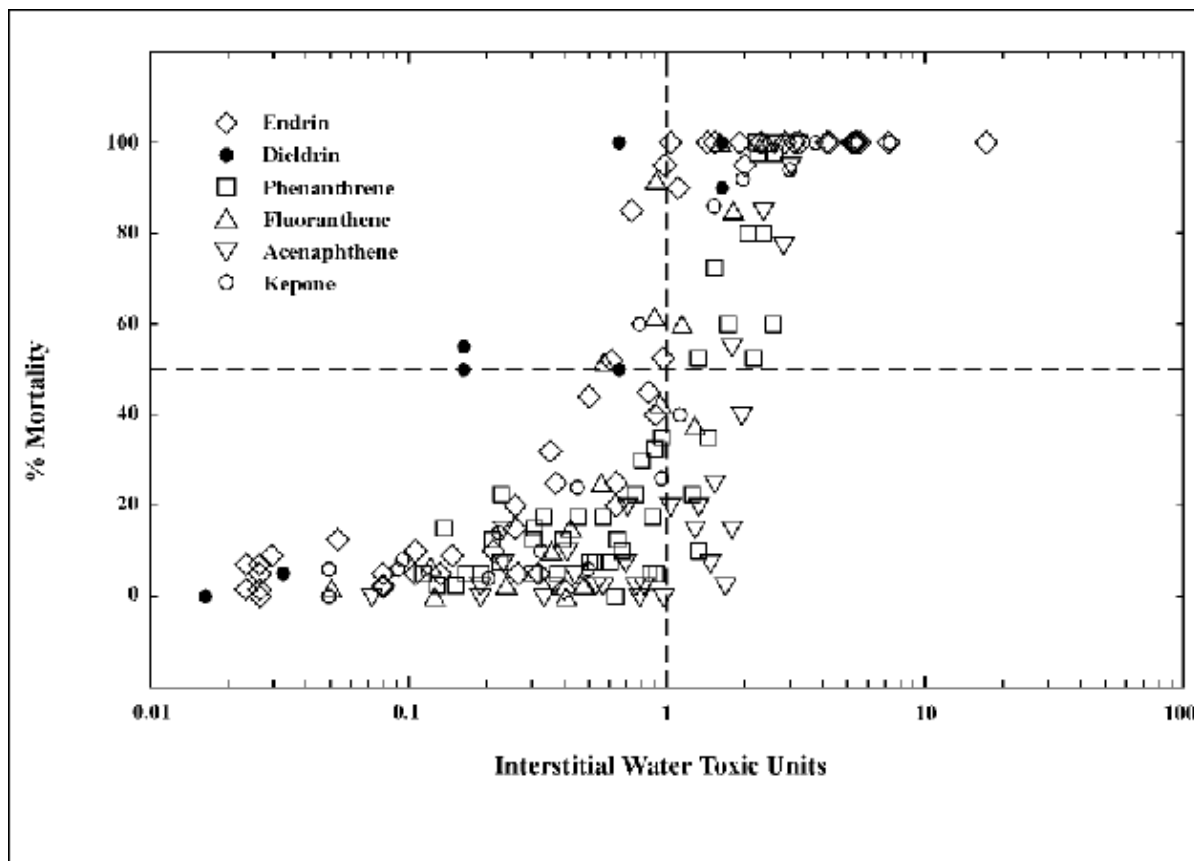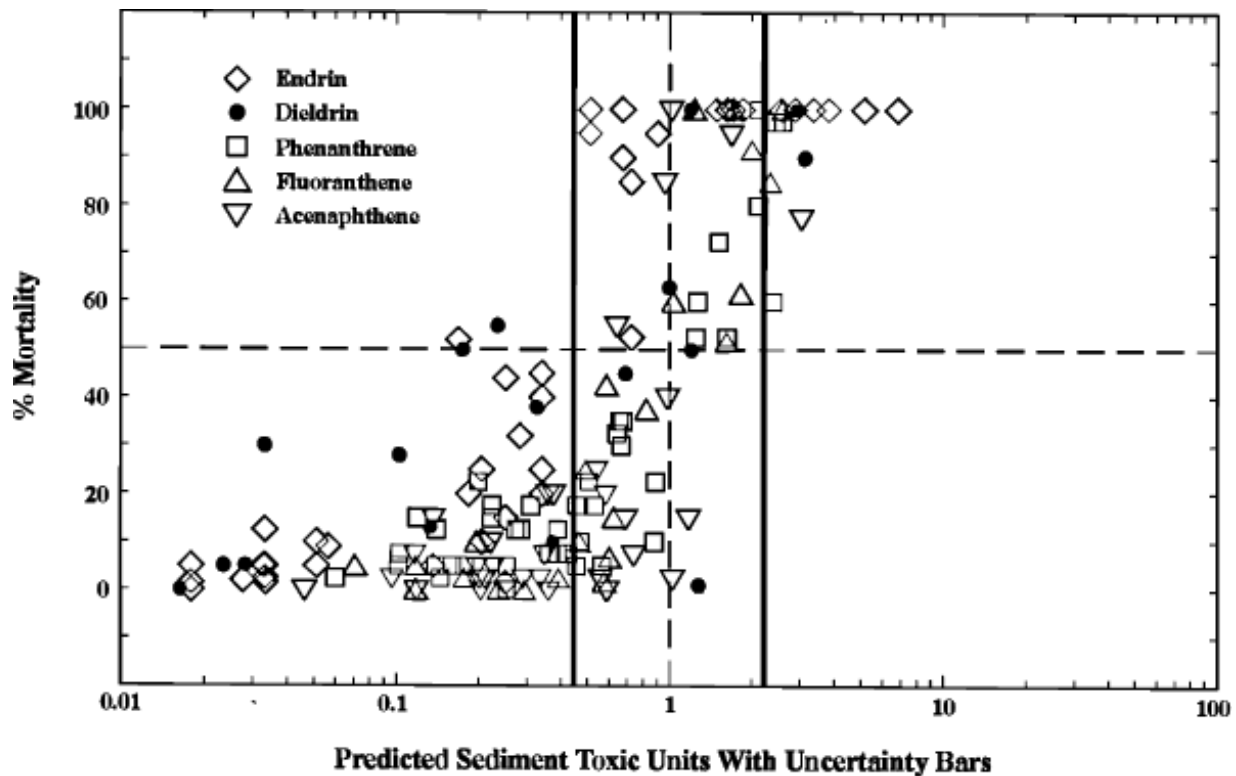**Figure 4-2.** Percent mortalities of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke et al., 1995), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and midge in sediments spiked with dieldrin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

Because DOC plays a significant role in the partitioning of dieldrin, the free interstitial water concentration is calculated using Equation 2-6 with the DOC values reported by Hoke and Ankley (1992) and the nominal interstitial water concentrations for Airport Pond sediments. The $\log_{10}K_{DOC}$ of 4.43 is taken from Kosian et al. (1995). This same approach was used for Pequaywan and West Bearskin Lakes data, with the poor results most likely due to the effects of DOC complexation (Hoke et al., 1995). Because only nominal interstitial water values are available, the dieldrin data shown in Figure 4-1 are presented to demonstrate the concept that interstitial water concentrations can be used to predict the response of an organism to a chemical that is not sediment specific.

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 one IWTU. At concentrations below one IWTU there should be less than 50% mortality, and at concentrations above one IWTU there should be greater

than 50% mortality. Figure 4-1 shows that, at concentrations below one IWTU, mortality was generally low and increased sharply at approximately one 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. This interstitial water normalization was not used to derive the ESB in this document because of the complexation of nonionic organic chemicals with interstitial water DOC (Section 2) and the difficulties of adequately sampling interstitial waters.

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 the ESB_{WQC}

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

Common Name, *Scientific Name*	Method,[a] Duration (days)	Water-only LC50 (μg/L)	Interstitial Water LC50 (μg/L)	TOC (%)	Dieldrin Sediment LC50s		Predicted[b] LC50 (μg/g$_{OC}$)	Ratio: Actual/ Predicted LC50	Reference
					Dry Wt. (μg/g)	OC (μg/g)			
Amphipod, *Hyalella azteca*	FT,M/10	7.3	54.3	1.7[c]	22.8	1,332[d]	1,391[e]	0.95	Hoke et al., 1995
Amphipod, *Hyalella azteca*	FT,M/10	7.3	236.1	2.9[c]	43.4	1,332[d]	1,391[e]	0.95	Hoke et al., 1995
Amphipod, *Hyalella azteca*	FT,M/10	7.3	491.6	8.7[c]	441.8	4,272[d]	1,391[e]	3.1	Hoke et al., 1995
Midge, *Chironomus tentans*	FT,M/10	1.1	0.50[f]	2.0[c]	1.5	95.3[d]	210[e]	0.45	Hoke et al., 1995
Midge, *Chironomus tentans*	FT,M/10	1.1	0.18[f]	1.5[c]	0.5	35.1[d]	210[e]	0.17	Hoke et al., 1995

[a]FT = flow-through; M = measured.
[b]Predicted LC50 (μg/g$_{OC}$) = water-only LC50 (μg/L) × K_{OC} (L/kg$_{OC}$) × 1 kg$_{OC}$/1000 g$_{OC}$; where $KOC = 10^{5.28}$.
[c]Mean reported TOC concentration.
[d]Calculated using individually measured TOC concentrations.
[e]Calculated using mean measured TOC concentrations.
[f]Interstitial water concentrations estimated from f_{OC}, K_{OC}, and measured sediment concentrations.

concentration 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 dieldrin in sediments were available for two benthic species. Data from tests with water column species were not included in this analysis. Testing of this component of the ESB_{WQC} derivation required three elements: (1) a water-only effect concentration, such as a 10-day LC50 value, in $\mu g/L$; (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 effects 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 dieldrin-spiked sediment tests with *H. azteca* (Hoke et al., 1995) were calculated (Table 4-2) using the $\log_{10} K_{OC}$ value of 5.28 from Section 2 of this document and the water-only LC50 value (7.3 $\mu g/L$). Ratios of actual to predicted sediment LC50 values for dieldrin averaged 1.4 (range 0.95 to 3.1) in tests with three sediments. Similarly, predicted sediment 10-day LC50 values for dieldrin-spiked sediment tests with *C. tentans* (Hoke et al., 1995) were calculated using the $\log_{10} K_{OC}$ of 5.28 and a 10-day water-only LC50 value of 1.1 $\mu g/L$ (Table 4-2). Ratios of predicted to actual sediment LC50 values for dieldrin averaged 0.28 (range 0.17 to 0.45) in tests with two sediments. The overall geometric mean ratio for both species was 0.73.

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 sediments ($\mu g/g_{OC}$) 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 one PSTU. Figure 4-2 shows that at concentrations below one PSTU mortality was generally low and increased sharply at one PSTU. Therefore, this comparison supports the concept that PSTUs 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 the effects concentration 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 $\pm 1.96 \times (ESB_{WQC}$ uncertainty). The value of 1.96 is the t statistic, which provides a 95% confidence interval around the ESB_{WQC}s.

Section 5

Benchmarks Derivation for Dieldrin

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 with 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 the 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 for the genera from eight or more families for which acceptable acute tests have been conducted on the substance. The EC50 represents the chemical concentration estimated to cause effects to 50% of the test organisms within a specified time period. 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 WQC Guidelines (Stephan et al., 1985). The FCV used in this document differs from the FCV in the dieldrin WQC document (U.S. EPA, 1980a) because it incorporates recent data not included in that document and omits some data that do not meet the data requirements of the 1985 WQC Guidelines.

The EqP method for calculating ESB$_{WQC}$s is 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$_{WQC}$ (μg/g sediment) are computed using the partition coefficient, K_P (L/g sediment), between sediment and interstitial water

$$\text{ESB}_{WQC} = K_P\,\text{FCV} \tag{5-1}$$

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 μg/g$_{OC}$) are

$$\text{ESB}_{WQCOC} = K_{OC}\,\text{FCV} \tag{5-2}$$

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 dieldrin ESB$_{WQC}$s.

Table 5-1. Equilibrium partitioning sediment benchmarks (ESB$_{WQC}$s) for dieldrin using the WQC FCV as the effect concentration

Type of Water Body	Log$_{10}K_{OW}$ (L/kg)	Log$_{10}K_{OC}$ (L/kg)	FCV (μg/L)	ESG$_{OC}$ (μg/g$_{OC}$)
Freshwater	5.37	5.28	0.06589	12[a]
Saltwater	5.37	5.28	0.1469	28[b]

[a]ESB$_{WQCOC}$ = ($10^{5.28}$ L/kg$_{OC}$) $\times$ (10^{-3} kg$_{OC}$/g$_{OC}$) $\times$ (0.06589 μg dieldrin/L) = 12 μg dieldrin/g$_{OC}$.
[b]ESB$_{WQCOC}$ = ($10^{5.28}$ L/kg$_{OC}$) $\times$ (10^{-3} kg$_{OC}$/g$_{OC}$) $\times$ (0.1469 μg dieldrin/L) = 28 μg dieldrin/g$_{OC}$.

The ESB$_{WQCOC}$ is applicable to sediments with f_{OC} ≥0.2%. For sediments with f_{OC} <0.2%, organic 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 the ESB$_{WQC}$s and field concentrations relative to identification of hot spots and the degree to which sediment concentrations do or do not exceed the ESB$_{WQC}$s values. The conversion from dry weight to organic carbon–normalized concentration can be done using the following formula

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

For example, a freshwater sediment with a concentration of 0.1 μg dieldrin/g$_{dry\,wt}$ and 0.5% TOC has an organic carbon-normalized concentration of 20 μg/g$_{OC}$ (= 0.1 μg/g$_{dry\,wt}$ × 100 ÷ 0.5), which exceeds the freshwater dieldrin ESB$_{WQC}$s of 12 μg/g$_{OC}$. Another freshwater sediment with the same concentration of dieldrin (0.1 μg/g$_{dry\,wt}$) but a TOC concentration of 5.0% would have an organic carbon–normalized concentration of 2.0 μg/g$_{OC}$ (= 0.1 μg/g$_{dry\,wt}$ × 100 ÷ 5.0), which is below the freshwater ESB$_{WQC}$s for dieldrin.

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, the ESB$_{WQCOC}$ may be "converted" to dry weight–normalized ESB$_{WQC}$ values (ESB$_{WQC\,dry\,wt}$). This "conversion" for each level of TOC is

$$\text{ESB}_{WQC\,dry\,wt} = \text{ESB}_{WQCOC} (\mu\text{g/g}_{OC}) \times (\% \text{ TOC} \div 100)$$

For example, the ESB$_{WQC\,dry\,wt}$ value for freshwater sediments with 1% organic carbon is 0.12 μg/g

$$\text{ESB}_{WQC\,dry\,wt} = 12\,\mu\text{g/g}_{OC} \times 1\% \text{ TOC} \div 100 = 0.12\,\mu\text{g/g}_{dry\,wt}$$

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 dieldrin ESB$_{WQC}$s can be estimated from the degree to which the available sediment toxicity data are explained 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 from sediments is equal on an organic carbon basis and (2) the effects concentration in sediment (μg/g$_{OC}$) can be estimated from the product of the effects concentration from water-only exposures (e.g., FCV (μg/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$_{WQC}$ (see Section 4.3 and the ESB Technical Basis Document). These freshwater and saltwater tests span a range of chemicals and organisms, they include both water-only and sediment exposures, and they are replicated within each chemical-organism-exposure media treatment. These data were analyzed using an analysis of variance (ANOVA) to estimate the uncertainty (i.e., the variance) associated with varying the 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 dieldrin are summarized in Appendix B. Only data from Hoke et al. (1995), as listed in Appendix B, were used in the uncertainty analysis because of mortality problems with *H. azteca* from Airport Pond as discussed in Sections 4.1 and 4.2. Data from Hoke and Ankley (1992), which used only Airport Pond sediments, have been used solely to compute partitioning. 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$_W$; μg/L) are related to the organic carbon–normalized LC50 values from sediment exposures (LC50$_{S,OC}$; μg/g$_{OC}$) via the partitioning equation

$$\text{LC50}_{S,OC} = K_{OC}\text{LC50}_W \tag{5-3}$$

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-only 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 of $\ln(LC50)$ for chemical–organism pair i. The error structure is assumed to be lognormal, 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,

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

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

The confidence limits are given in Table 5-3.

The ESB_{WQCOC} are applicable to sediments with $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic carbon normalization and $ESB_{WQC}s$ do not apply.

Table 5-2. Analysis of variance for derivation of confidence limits of the ESBs for dieldrin

Source of Uncertainty	Parameter	Value ($\mu g/g_{oc}$)
Exposure media	σ_α	0.41
Replication	σ_ϵ	0.29
ESG Sediment Guideline	σ_{ESG} [a]	0.41

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

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

Type of Water Body	ESG_{OC} ($\mu g/g_{oc}$)	95% Confidence Limits ($\mu g/g_{oc}$)	
		Lower	Upper
Freshwater	12	5.4	27
Saltwater	28	12	62

5.3 Comparison of Dieldrin ESB 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 dieldrin and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figures 5-1 and 5-2). The effect concentrations in sediments are predicted from water-only toxicity data and K_{OC} values (see Section 4). Chronically acceptable concentrations are extrapolated from GMAVs from water-only, 96-hour lethality tests using the FACR. These two predictive

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 dieldrin ESB$_{WQC}$ ± 95% confidence limits. Solid symbols are benthic genera; open symbols are water column genera. Sediment 10-day LC50$_{S,OC}$ values (calculated from Hoke et al., 1995; see Table 4-1) for the amphipods *C. tentans* (H) and *H. azteca* (✳) are provided for comparison. Error bars around the LC50$_{S,OC}$ values indicate the observed range of LC50s.**

values are used to estimate chronically acceptable sediment concentrations (predicted genus mean chronic value, PGMCV) for dieldrin from GMAVs (Appendix A), the FACR (Table 3-2), and the K_{OC} (Table 5-1)

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

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 dieldrin (Table 4-1;

Appendix B) are placed in the PGMCV rank appropriate to the test-specific effect concentration. For example, the mean 10-day $LC50_{S,OC}$ for *C. tentans*, 57.8 $\mu g/g_{OC}$, is placed between the PGMCV of 25.0 $\mu g/g_{OC}$ for the stonefly, *Claassenia*, and the PGMCV of 153 $\mu g/g_{OC}$ for the fish, *Micropterus*. 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, organism sensitivities, and PGMCVs is not intended to provide accurate predictions of the responses of taxa or communities of
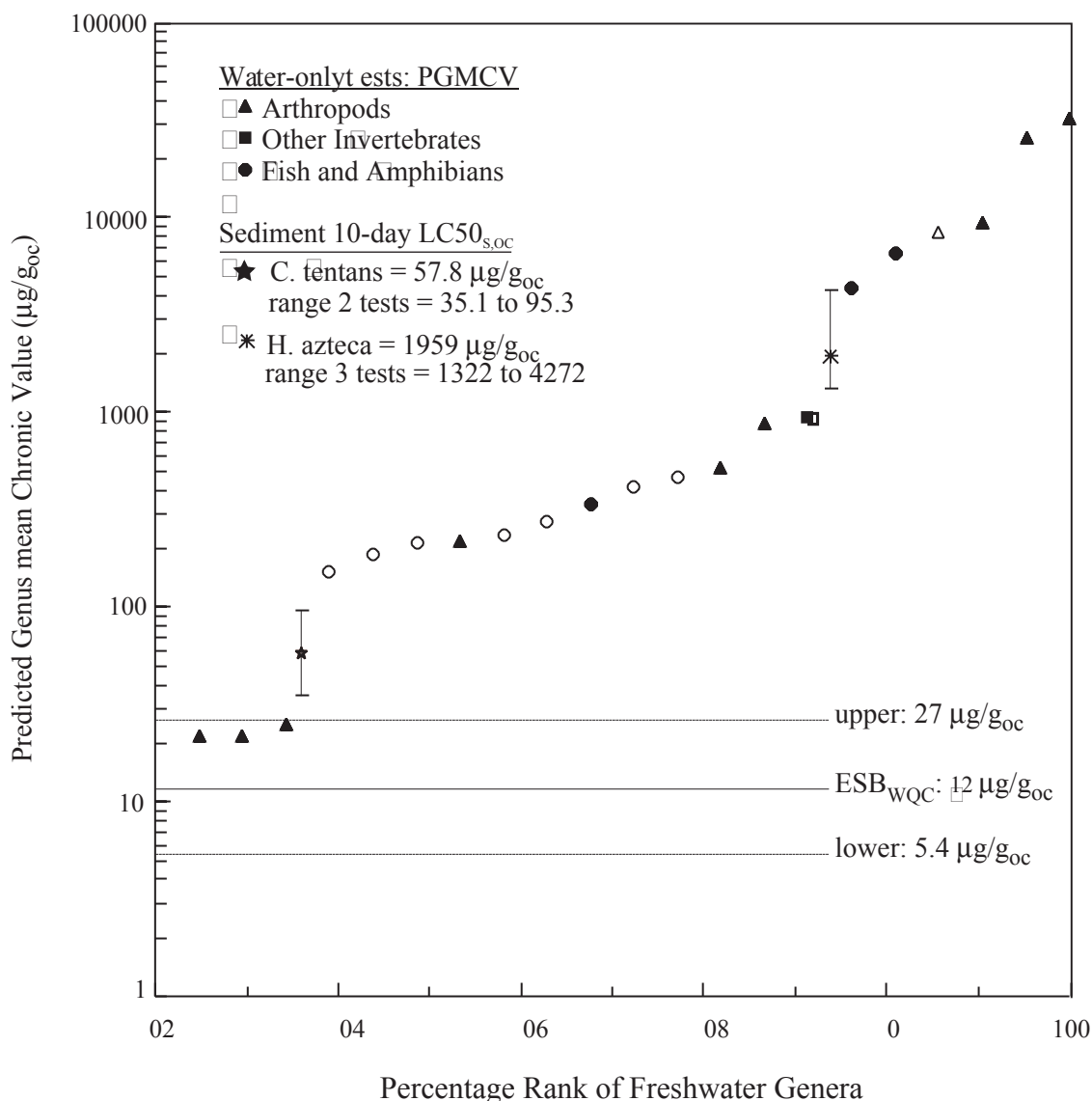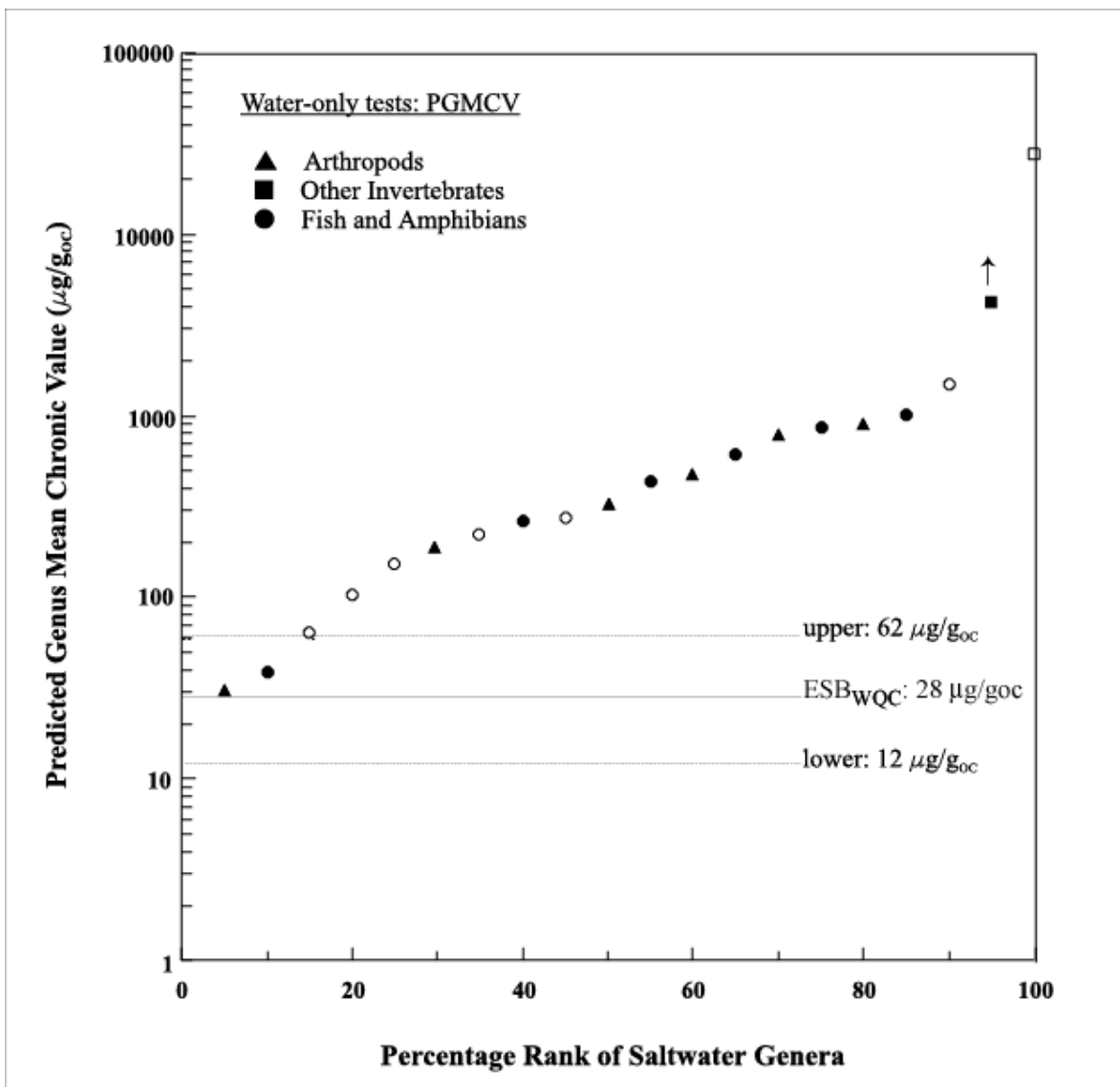

Figure 5-2. Predicted genus mean chronic values (PGMCV) calculated from water-only toxicity values (Equation 5-7; Appendix A) using saltwater species versus percentage rank of their sensitivity. Solid symbols are benthic genera; open symbols are water column genera. Arrows indicate greater than values.

benthic organisms relative to specific concentrations of dieldrin 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 dieldrin.

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 dieldrin (12 μg/g$_{OC}$) is less than any of the PGMCVs or LC50 values from spiked sediment toxicity tests (Figure 5-1). The PGMCVs for 18 of 21 freshwater genera are greater than the upper 95% confidence interval of the ESB$_{WQC}$ (27 μg/g$_{OC}$). The PGMCVs for the stoneflies *Pteronarcella* (22 μg/g$_{OC}$), *Pteronarcys* (22 μg/g$_{OC}$), and *Claassenia* (26 μg/g$_{OC}$) 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 above the ESB$_{WQC}$ and possibly less than the 95% upper confidence interval. For dieldrin, PGMCVs range over three orders of magnitude from the most sensitive to the most tolerant genus (Figure 5-1). A sediment concentration 20 times the ESB$_{WQC}$ would include the PGMCVs of 4 of the 13 benthic genera tested including stoneflies, isopods, and fish.

Tolerant benthic genera such as the amphipod *Gammarus* and the crayfish *Orconectes* may not be chronically affected in sediments with dieldrin concentrations up to 1,000 times the ESB$_{WQC}$ (Figure 5-1; Appendix A). Data from lethality tests with freshwater organisms exposed to dieldrin-spiked sediments substantiates this projection; the 10-day LC50 values from three tests with the amphipod *H. azteca* ranged from 110 to 360 times the ESB$_{WQC}$ of 12 μg/g$_{OC}$, the 10-day LC50s from two tests with the midge *C. tentans* ranged from 2.9 to 7.9 times the ESB$_{WQC}$ (see insert Figure 5-1; corresponding values from Table 4-1).

The saltwater ESB$_{WQC}$ for dieldrin (28 μg/g$_{OC}$) is less than all of the PGMCVs for saltwater genera (Figure 5-2). The PGMCVs for the penaeid shrimp *Penaeus duorarum* (31 μg/g$_{OC}$) and the fish *Anguilla rostrata* (39 μg/g$_{OC}$) are lower than the upper 95% confidence interval for the ESB$_{WQC}$ (62 μg/g$_{OC}$). For dieldrin, PGMCVs from the most sensitive to the most tolerant saltwater genus range over two orders of magnitude. A

sediment concentration 17 times the ESB$_{WQC}$ would include the PGMCVs of 7 of the 13 benthic genera tested including 4 arthropod and 3 fish genera. Other genera of benthic arthropods, polychaetes, and fishes are less sensitive and might not be expected to be chronically affected in sediments with dieldrin concentrations 30 times the ESB$_{WQC}$.

5.4 Comparison of Dieldrin ESB$_{WQC}$s to STORET, National Status and Trends, and Corps of Engineers, San Francisco Bay Databases for Sediment Dieldrin

Dieldrin is frequently measured when samples are taken to measure sediment contamination, and dieldrin 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 dieldrin benchmarks. In order to investigate this possibility, the dieldrin benchmarks were compared with data from several available databases of sediment chemistry.

The following description of dieldrin distributions in Figure 5-3 is somewhat misleading because it includes data from samples in which the dieldrin 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 artificial upper bounds, not measured values, the percentage of samples in which the ESB$_{WQC}$ values were actually exceeded may be less than the percentage reported. 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 dieldrin in the sediments of the nation's water bodies. Log probability plots of dieldrin concentrations on a dry weight basis in sediments are shown in Figure 5-3. Dieldrin was found at varying concentrations in sediments from rivers, lakes, and near-coastal water bodies in the United States. This was because of its widespread use and quantity applied during the 1960s and early 1970s. It was restricted from registration and production in the United States in 1974. Median concentrations were generally at or near detection limits in most water bodies for data after 1986. There was significant variability with dieldrin concentrations in sediments ranging over nine orders of magnitude within the country.
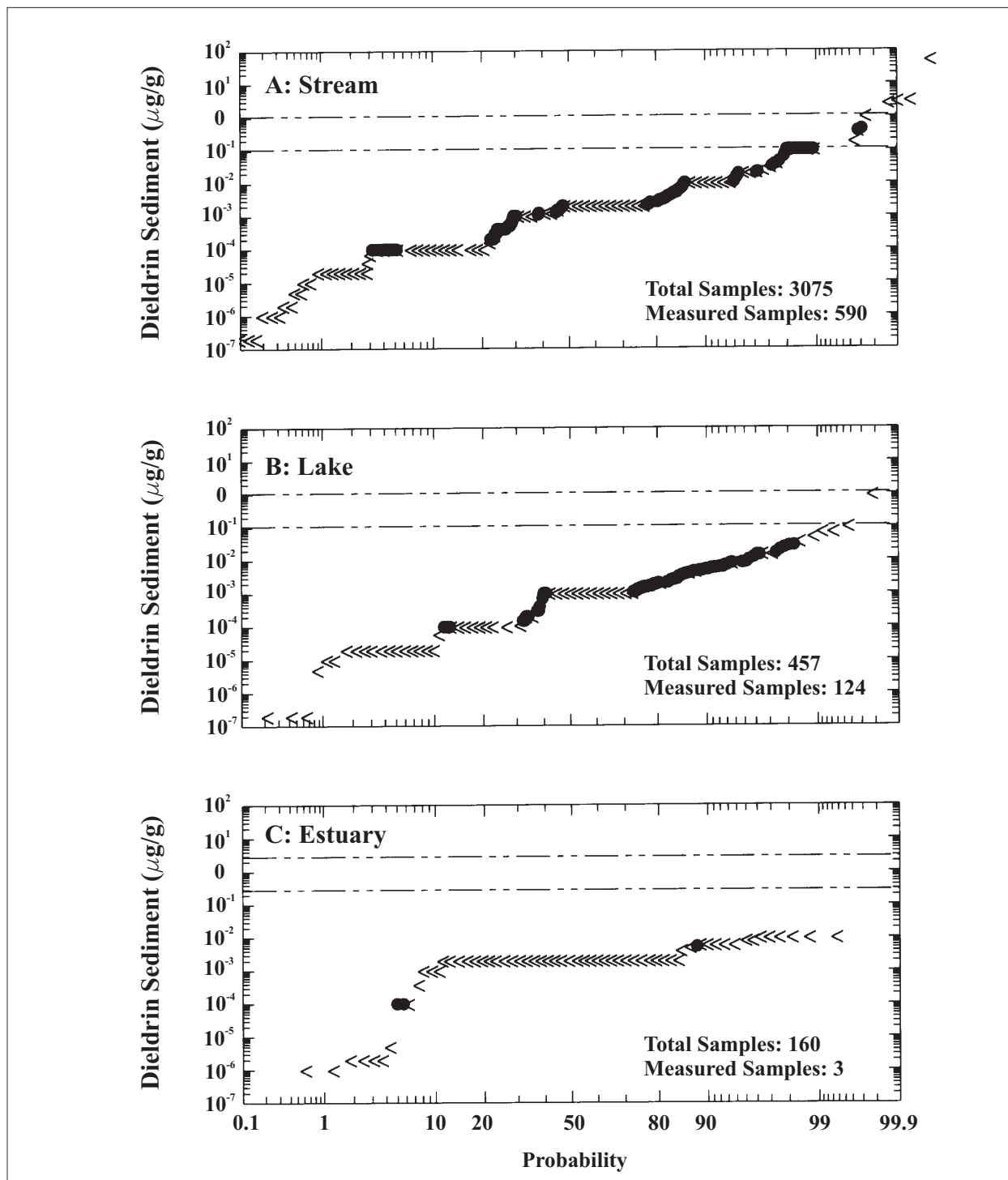
Figure 5-3. Probability distribution of concentrations of dieldrin in sediments from streams (A), lakes (B), and estuaries (C) in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared with the dieldrin ESB$_{WQC}$ values. Sediment dieldrin concentrations less than the detection limits are shown as less than symbols (<); measured concentrations are shown as solid circles (l). 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%.

The ESB$_{WQC}$s for dieldrin can be compared to existing concentrations of dieldrin in sediments of natural water systems in the United States as contained in the STORET database (U.S. EPA, 1989b). These data are 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 dieldrin's distribution in sediments as examples only. For freshwater sediments, ESB$_{WQC}$ concentrations were 0.12 μg/g dry weight in sediments having 1% organic carbon and 1.2 μg/g dry weight in sediments having 10% organic carbon; for marine sediments, ESB$_{WQC}$s were 0.28 μg/g dry weight and 2.8 μg/g dry weight, respectively. Figure 5-3 presents comparisons of these ESB$_{WQC}$s with probability distributions of observed sediment dieldrin levels for streams and lakes (freshwater systems, shown on A and B) and estuaries (marine systems, C).

For both streams (n=3,075) and lakes (n=457), the ESB$_{WQC}$s of 0.12 μg/g dry weight for 1% organic carbon freshwater sediments and of 1.2 μg/g dry

weight for 10% organic carbon freshwater sediments were exceeded in less than 1% of the samples. In estuaries, the data (n=160) indicate that neither benchmark, 0.28 μg/g dry weight for sediments having 1% organic carbon nor 2.8 μg/g dry weight for sediments having 10% organic carbon, was exceeded by the post 1986 samples. Concentrations of dieldrin in sediments from estuaries were two orders of magnitude below the ESB$_{WQC}$ value for 1% organic carbon sediments and three orders of magnitude below the ESB$_{WQC}$ value for sediments with TOCs of 10%.

A second database developed as part of the National Status and Trends Program (NOAA, 1991) was available for assessing contaminant levels in marine sediments that were representative of areas away from sources of contamination. The probability distribution for these data, on an organic carbon basis, was compared with the saltwater ESB$_{WQC}$ for dieldrin (28 μg/g$_{OC}$) in Figure 5-4. Data presented were from sediments with 0.20% to 31.9% organic carbon. The median organic carbon–normalized dieldrin concentration (0.080 μg/g$_{OC}$) was two orders of
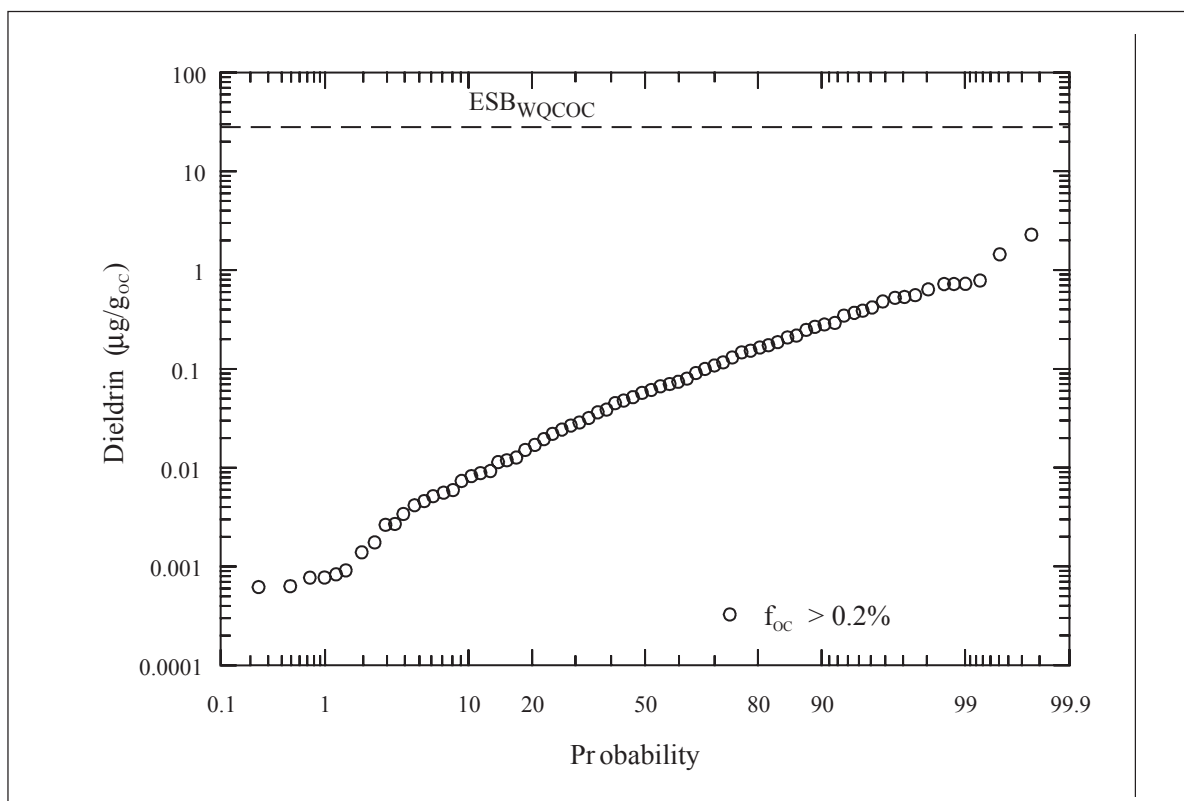
Figure 5-4. **Probability distribution of concentrations of dieldrin in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal dashed line is the saltwater ESB$_{WQC}$ value of 28 µg/g$_{OC}$.**

magnitude below the ESB_{WQC} of 28 $\mu g/g_{OC}$. None of these samples (n=408) exceeded the benchmarks. Hence, these results are consistent with the preceding comparison between the marine ESB_{WQC} and STORET data.

A third set of data has been 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 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 dieldrin 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 dieldrin in sediments were normalized by the organic carbon content, and the results are displayed as a probability plot in Figure 5-5

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 actual measurements were obtained were at least an order of magnitude lower than the ESB_{WQC}. An estimate of the possible frequency distribution of sediment concentrations of dieldrin was developed by the application of an analysis technique that accounts for the varying detection limits and the presence of nondetected observations (El-Shaarawi 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 the pesticide in relation to the sediment benchmarks.

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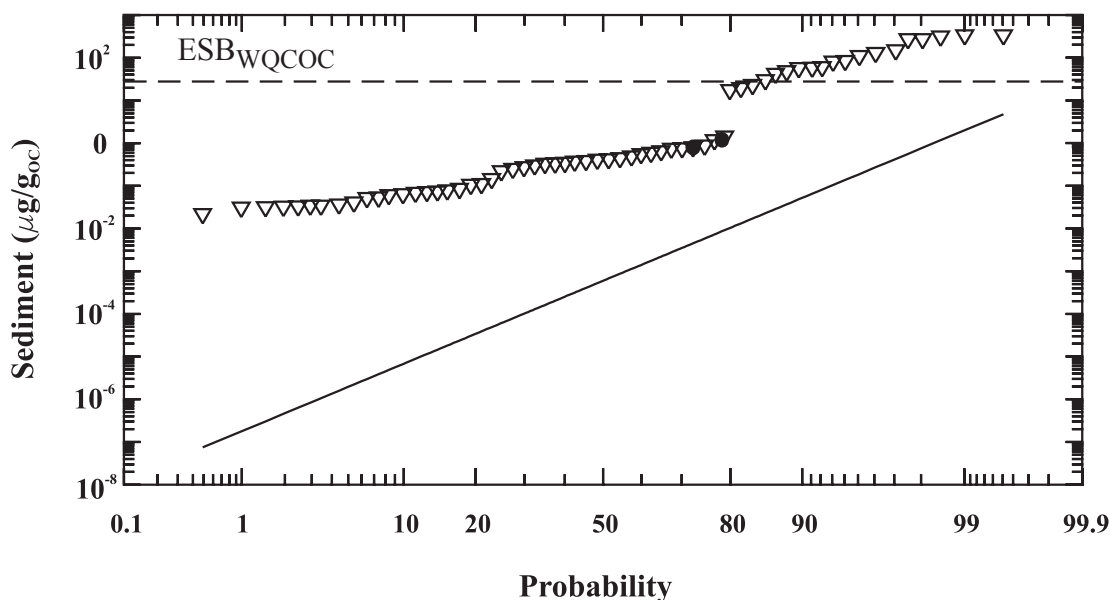
Figure 5-5.   Probability distribution of organic carbon–normalized sediment dieldrin concentrations from the U.S. Army Corps of Engineers (1991) monitoring program of San Francisco Bay. Sediment dieldrin concentrations less than the detection limits are shown as open triangles (∇); measured concentrations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

concentrations and the extent to which they may exceed the ESB$_{WQC}$.

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 represents 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 applying EqP-derived 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 dieldrin 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 under-protective 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 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 dieldrin where dieldrin concentrations in sediment are below the ESB_{WQC} values of 12 µg dieldrin/g_{OC} for freshwater sediments and 28 µg dieldrin/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 5.4 to 27 µg/g_{OC} for freshwater sediments and 12 to 62 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 dieldrin 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 dieldrin 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 dieldrin; it does not account for potential antagonistic, additive, or synergistic effects that may occur in sediments containing a mixture of dieldrin 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 dieldrin may show additive toxicity with dieldrin

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 dieldrin 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 dieldrin 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 dieldrin 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 dieldrin. Predicted concentrations of dieldrin 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 dieldrin where dieldrin concentrations in sediment are below the ESB_{WQC} values of 12 µg dieldrin/g_{OC} for freshwater sediments and 28 µg dieldrin/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 dieldrin or the potential for bioaccumulation and trophic transfer of dieldrin 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

References

Adams WJ, Kimerle RA, Mosher RG. 1985. Aquatic safety assessment of chemicals sorbed to sediments. In Cardwell RD, Purdy R, Bahner RC, eds, *Aquatic Toxicology and Hazard Assessment: Seventh Symposium*. STP 854. American Society for Testing and Materials, Philadelphia, PA, pp 429–453.

Adema DMM. 1978. *Daphnia magna* as a test animal in acute and chronic toxicity tests. *Hydrobiol* 59:125–134.

Adema DMM, Vink GJ. 1981. A comparative study of the toxicity of 1,1,2-trichloroethane, dieldrin, pentachlorophenol, and 3,4 dichloroaniline for marine and fresh water. *Chemosphere* 10:533–554.

Alabaster JS, Lloyd R, eds. 1982. Mixtures of toxicants. In *Water Quality Criteria for Freshwater Fish*. Butterworth Scientific, London, UK.

Betsill JD. 1990. The sorption of hydrophobic organic compounds in the presence of environmental concentrations of dissolved humic and fulvic acids at variable pH values. PhD thesis. Oklahoma State University, Stillwater, OK.

Briggs GG. 1981. Theoretical and experimental relationships between soil adsorption, octanol-water partition coefficients, water solubilities, bioconcentration factors and the parachlor. *J Agric Food Chem* 29:1050–1059.

Brooke DN, Dobbs AJ, Williams N. 1986. Octanol-water partition coefficients (P): Measurements, estimation and interpretation, particularly for chemicals with P>10^5. *Ecotoxicol Environ Saf* 11:251–260.

Brooke LT. 1993a. Conducting toxicity tests with freshwater organisms exposed to dieldrin, fluoranthene and phenanthrene. Final Report. Environmental Research Laboratory, U.S. Environmental Protection Agency, Duluth, MN.

Brooke LT. 1993b. Acute and chronic toxicity testing of several pesticides to five species of aquatic organisms. Final Report. Environmental Research Laboratory, U.S. Environmental Protection Agency, Duluth, MN.

Burkhard LP. 2000. Estimating dissolved organic carbon partition coefficients for nonionic organic chemicals. *Environ Sci Technol* 34:4663-4668.

Cairns J Jr, Loos JJ. 1966. Changes in guppy populations resulting from exposure to dieldrin. *Prog Fish Cult* 28:220–226.

Chadwick GG, Kiigemagi U. 1968. Toxicity evaluation of a technique for introducing dieldrin into water. *J Water Pollut Control Fed* 40:76–82.

Chadwick GG, Shumway DL. 1969. Effects of dieldrin on the growth and development of steelhead trout. *Proceedings*, The Biological Impact of Pesticides in the Environment, Corvallis, OR, August 18–20, 1969, pp 90–96.

Chapman GA. 1987. Establishing sediment criteria for chemicals—Regulatory perspective. In Dickson KL, Maki AW, Brungs WA, eds, *Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems*. Pergamon Press, New York, NY, pp 355–376.

Davis HC, Hidu H. 1969. Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. *Fisheries Bull* 67:393–404.

De Bruijn J, Busser F, Seinen W, Hermens J. 1989. Determination of octanol/water partition coefficients for hydrophobic organic chemicals with the slow-stirring method. *Environ Toxicol Chem* 8:499–512.

De Kock AC, Lord DA. 1987. A simple procedure for determining octanol-water partition coefficients using reverse phase high performance liquid chromatography (RPHPLC). *Chemosphere* 16:133–142.

DeWitt TH, Ozretich RJ, Swartz RC, Lamberson JO, Shults DW, Ditsworth GR, Jones JKP, Hoselton L, Smith LM. 1992. The influence of organic matter quality on the toxicity and partitioning of sediment-associated fluoranthene. *Environ Toxicol Chem* 11:197–208.

Di Toro DM. 1985. A particle interaction model of reversible organic chemical sorption. *Chemosphere* 14:1503–1538.

Di Toro DM, Zarba CS, Hansen DJ, Berry WJ, Swartz RC, Cowan CE, Pavlou SP, Allen HE, Thomas NA, Paquin PR. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. *Environ Toxicol Chem* 10:1541–1583.

Douglas MT, Chanter DO, Pell IB, Burney GM. 1986. A proposal for the reduction of animal numbers required for the acute toxicity to fish test (LC50 determination). *Aquat Toxicol* 8:243–249.

Earnest RD, Benville PE Jr. 1972. Acute toxicity of four organo-chlorine insecticides to two species of surf perch. *Calif Fish Game* 58:127–132.

Eisler R. 1969. Acute toxicities of insecticides to marine decapod crustaceans. *Crustaceana* 16:302–310.

Eisler R. 1970a. Factors affecting pesticide-induced toxicity in an estuarine fish. Technical Paper 45. Bureau of Sport Fisheries and Wildlife, U.S. Department of the Interior, Washington, DC.

Eisler R. 1970b. Acute toxicities of organic organochlorine and organophosphorous insecticides to estuarine fishes. Technical Paper 46. Bureau of Sport Fisheries and Wildlife, U.S. Department of the Interior, Washington, DC.

Ellington JJ, Stancil FE Jr. 1988. Octanol/water partition coefficients for evaluation of hazardous waste land disposal: Selected chemicals. EPA/600/M-88/010. Environmental Research Brief. Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.

El-Shaarawi AH, Dolan DM. 1989. Maximum likelihood estimation of water quality concentrations from censored data. *Can J Fish Aquat Sci* 46:1033–1039.

Eye JD. 1968. Aqueous transport of dieldrin residues in soils. *J Water Pollut Control Fed* 40:R316–R332.

Grathwohl P. 1990. Influence of organic matter from soils and sediments from various origins on the sorption of some chlorinated aliphatic hydrocarbons: Implications on K_{OC} correlations. *Environ Sci Technol* 24:1687–1693.

Gschwend PM, Wu S-c. 1985. On the constancy of sediment-water partition coefficients of hydrophobic organic pollutants. *Environ Sci Technol* 19:90-96.

Hartley D, Kidd H, eds. 1987. *The Agrochemicals Handbook. 2nd Ed.* Royal Society of Chemistry, University of Nottingham, England.
Henderson C, Pickering QH, Tarzwell CM. 1959. Relative toxicity of ten chlorinated hydrocarbon insecticides to four species of fish. *Trans Am Fish Soc* 88:23–32.

Hoke RA, Ankley GT, 1992. Results of Airport Pond dieldrin-spiked sediments. Memorandum to D. Hansen and D. Di Toro, HydroQual, Inc., Mahwah, NJ, January 27, 1992. 8 pp.

Hoke RA, Kosian PA, Ankley GT, Cotter AM, Vandermeiden FM, Phipps G Sanchez-Camazano M. 1997. Effect of the nature of exogenous organic matter on pesticide sorption by the soil. *Arch Environ Contam Toxicol* 33:117–124.

Karickhoff SW, Carreira LA, Melton C, McDaniel VK, Vellino AN, Nute DE. 1989. Computer prediction of chemical reactivity—the ultimate SAR. EPA/600/M-89/017. Environmental Research Brief. Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.

Karickhoff SW, McDaniel VK, Melton C, Vellino AN, Nute DE, Carreira LA. 1991. Predicting chemical reactivity by computer. Environ Toxicol Chem 10:1405-1416.

Karickhoff SW, Long JM. 1995. Internal report on summary of measured, calculated, and recommended log K_{OW} values. Internal Report. Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.

Karickhoff SW, Long JM. 1996. Protocol for setting K_{OW} values. Internal Report. Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, GA.

Katz M. 1961. Acute toxicity of some organic insecticides to three species of salmonids and to the threespine stickleback. *Trans Am Fish Soc* 90:264–269.

Korn S, Earnest RD. 1974. Acute toxicity of twenty insecticides to striped bass, *Morone saxatilis*. *Calif Fish Game* 60:128–131.

Kosian PA, Hoke RA, Ankley GT, Vandermeiden FM. 1995. Determination of dieldrin binding to dissolved organic material in sediment interstitial water using a reverse-phase separation technique. *Environ Toxicol Chem* 14:445–450.

Mabey WR, Smith JH, Podoll RT, Johnson HL, Mill T, Chou TW, Gates J, Partridge IW, Jaber H, Vandenberg D. 1982. Aquatic fate process data for organic priority pollutants. EPA-440/4-81-014. Final Report. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, DC.

Macek KJ, Hutchinson C, Cope OB. 1969. The effects of temperature on the susceptibility of bluegills and rainbow trout to selected pesticides. *Bull Environ Contam Toxicol* 4:174–183.

Mackay D, Powers B. 1987. Sorption of hydrophobic chemicals from water: A hypothesis for the mechanism of the particle concentration effect. *Chemosphere* 16:745–757.

Mayer FL, Ellersieck MR. 1986. Manual of acute toxicity: Interpretation and database for 410 chemicals and 66 species of freshwater animals. Resource Publication 160. Fish and Wildlife Service, U.S. Department of the Interior, Washington DC.

McCauley DJ. 1997. Acute dieldrin toxicity testing: acute and early life-stage toxicity testing of dieldrin with fathead minnow (*Pimephales promelas*). Final Report. Office of Science and Technology, U.S. Environmental Protection Agency, Washington, DC.

McLeese DW, Metcalfe CD. 1980. Toxicities of eight organochlorine compounds in sediment and seawater to *Crangon septemspinosa*. *Bull Environ Contam Toxicol* 25:921–928.

McLeese, DW, Burridge LE, Dinter DJ. 1982. Toxicities of five organochlorine compounds in water and sediment to *Nereis virens*. *Bull Environ Contam Toxicol* 8:216–220.

National Academy of Sciences (NAS). 1973. Water Quality Criteria, 1972. EPA-R3-73-033. National Academy of Sciences, U.S. Environmental Protection Agency, Washington, DC.

National Oceanic and Atmospheric Administration (NOAA). 1991. National Status and Trends Program—Second summary of data on chemical contaminants in sediments from the National Status and Trends Program. Technical Memorandum NOS OMA 59. Office of Oceanography and Marine Assessment, Rockville, MD.

Nebeker AV, Schuytema GS, Griffis WL, Barbitta JA, Carey LA. 1989. Effect of sediment organic carbon on survival of *Hyalella azteca* exposed to DDT and endrin. *Environ Toxicol Chem* 8:705–718.

Noreen EW. 1989. *Computer Intensive Methods for Testing Hypotheses: An Introduction*. John Wiley, New York, NY.

Parrish PR, Couch JA, Forester J, Patrick JM Jr, Cook GH. 1973. Dieldrin: Effects on several estuarine organisms. *Proceedings*, 27th Annual Conference S.E. Association Game Fish Commission, Hot Springs, AK, October 14–17, 1973, pp 427–434.

Roelofs TD. 1971. Effects of dieldrin on the intrinsic rate of increase of the guppy *Poecilia reticulata* Peters. PhD thesis. Oregon State University, Corvallis, OR.

Sanders HO. 1969. Toxicity of pesticides to the crustacean, *Gammarus lacustris*. Technical Paper No. 25. Bureau of Sport Fisheries and Wildlife, U.S. Department of the Interior, Washington, DC.

Sanders HO. 1972. Toxicity of some insecticides to four species of malacostracan crustaceans. Technical Paper No. 66. Bureau of Sport Fisheries and Wildlife, U.S. Department of the Interior, Washington, DC.

Sanders HO, Cope OB. 1966. Toxicities of several pesticides to two species of cladocerans. *Trans Am Fish Soc* 95:165–169.

Schoettger RA. 1970. Fish-Pesticide Research Laboratory, Progress in Sport Fishery Research. Research Publication 106. Bureau of Sport Fisheries and Wildlife, U.S. Department of the Interior, Washington, DC.

Schuytema GA, Nebeker AV, Griffis WL, Miller CE. 1989. Effects of freezing on toxicity of sediments contaminated with DDT and endrin. *Environ Toxicol Chem* 8:883–891.

Sharom MS, Miles JR, Harris CR, McEwen FL. 1980. Persistence of 12 insecticides in water. *Water Res* 14:1089–1093.

Stephan CE, Mount DI, Hansen DJ, Gentile JH, Chapman GA, Brungs WA. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. PB85-227049. National Technical Information Service, Springfield, VA.

Suter SW, Mabrey JM. 1994. Toxicological benchmarks for screening potential contaminants of concern for effects on aquatic biota: 1994 revision. ES/ER/TM-96/RI. Oak Ridge National Laboratory, Environmental Sciences Division. Oak Ridge, TN.

Swartz RC. 1991. Acenaphthene and phenanthrene files. Memorandum to D. Hansen, HydroQual, Inc., Mahwah, NJ, June 26, 1991. 160 pp.

Swartz RC, Schults DW, DeWitt TH, Ditsworth GR, Lamberson JO. 1990. Toxicity of fluoranthene in sediment to marine amphipods: A test of the equilibrium-partitioning approach to sediment quality criteria. *Environ Toxicol Chem* 9:1071–1080.

Tarzwell CM, Henderson C. 1957. Toxicity of dieldrin to fish. *Trans Am Fish Soc* 86:245–257.

U.S. Army Corps of Engineers (COE). 1991. Monitoring Program for San Francisco Bay Sediments. 1988 to 1990. Memorandum to D. Di Toro, HydroQual, Inc., Mahwah, NJ, 1991.

U.S. Environmental Protection Agency. 1980a. Ambient water quality criteria aldrin/dieldrin. EPA 440/5-80-019. Office of Water Regulations and Standards, Washington, DC.

U.S. Environmental Protection Agency. 1980b. Unpublished laboratory data. Environmental Research Laboratory, Gulf Breeze, FL.

U.S. Environmental Protection Agency. 1985. Appendix B—Response to public comments on "Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses." July 19, 1985. Federal Register 50:30793–30796.

U.S. Environmental Protection Agency. 1987a. Quality criteria for water, 1986. EPA 440/5-86-001. Office of Water Regulations and Standards, Washington, DC. U.S. Environmental Protection Agency. 1987b. Acute toxicity handbook of chemicals to estuarine organisms. EPA/600/8-87/017. Environmental Research Laboratory, Gulf Breeze, FL.

U.S. Environmental Protection Agency. 1989a. Sediment classification methods compendium. PB92-231679. National Technical Information Service, Springfield, VA.

U.S. Environmental Protection Agency. 1989b. Handbook: Water quality control information system, STORET. Office of Water and Hazardous Materials, Washington, DC.

U.S. Environmental Protection Agency. 1999. National recommended water quality criteria—Correction. EPA-822-2-99-001. April 1999. Washington, DC.

U.S. Environmental Protection Agency. 2000. Comment response document for the proposed equilibrium partitioning sediment guidelines for the protection of benthic organisms. Office of Water, Office of Science and Tecnhology, Washington, DC. (draft)

U.S. Environmental Protection Agency. 2003a. Technical basis for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Nonionic organics. EPA-600-R-02-014. Office of Research and Development, Washington, DC. (draft)

U.S. Environmental Protection Agency. 2003b. Procedures for the derivation of site-specific equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Nonionic organics. EPA-600-R-02-012. Office of Research and Development, Washington, DC. (draft)

U.S. Environmental Protection Agency. 2003c. 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.

U.S. Environmental Protection Agency. 2003d. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Metal mixtures (cadmium, copper, lead, nickel, silver, and zinc). EPA-600-R-02-011. Office of Research and Development, Washington, DC. (draft)

U.S. Environmental Protection Agency. 2002e. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: PAH mixtures. EPA-600-R-02-013. Office of Research and Development, Washington, DC. (draft)

U.S. Environmental Protection Agency. 2003f. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Nonionics compendium. EPA-822-R-00-06. EPA-600-R-02-016. Office of Research and Development, Washington, DC. (draft)

Wang YS. 1988. The contamination and bioconcentration of aldrin, dieldrin and endrin in lower lakes at Rocky Mountain Arsenal. PhD thesis. Colorado State University, Fort Collins, CO.

Xing B, McGill WB, Dudas MJ. 1994. Cross-correlation of polarity curves to predict partition coefficients of nonionic organic contaminants. *Environ Sci Technol* 28:1929-1933.

Appendix A

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

Appendix A

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
						LC50/EC50[e] (µg/L)			

Freshwater Species

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	Test	Species[f]	Genus[g]	Overall GMAV[h]	Reference
Annelid, *Lumbriculus variegatus*	A	I	FT	M	21.8	21.8	21.8	21.8	Brooke, 1993b
Cladoceran, *Daphnia pulex*	J	W	S	U	250	—	—	—	Mayer and Ellersieck, 1986
Cladoceran, *Daphnia pulex*	X	W	S	U	250	250	—	—	Sanders and Cope, 1966
Cladoceran, *Daphnia magna*	J	W	S	U	>200	—	—	—	Adema, 1978
Cladoceran, *Daphnia magna*	A	W	S	U	>200	—	—	—	Adema, 1978
Cladoceran, *Daphnia magna*	J	W	R	M	79.5	>147.1	>191.8	>191.8	Brooke, 1993a
Cladoceran, *Simocephalus serrulatus*	J	E,W	S	U	240	—	—	—	Sanders and Cope, 1966; Mayer and Ellersieck, 1986
Cladoceran, *Simocephalus serrulatus*	X	E,W	S	U	190	213.5	213.5	213.5	Sanders and Cope, 1966; Mayer and Ellersieck, 1986
Isopod, *Asellus brevicaudus*	X	E	S	U	5.0	5.0	5.0	5.0	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	A	E	S	U	640	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	X	E	S	U	600	619.7	—	—	Sanders, 1972; Mayer and Ellersieck, 1986

A-2

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	Test	HMAV Species[f]	Genus[g]	Overall GMAV[h]	Reference
						LC50/EC50[e] (μg/L)			
Scud, *Gammarus lacustris*	X	E	S	U	567.5	567.5	593.0	593.0	Sanders, 1969
Glass shrimp, *Palaemonetes kadiakensis*	X	E	S	U	20	20	20	20	Sanders, 1972
Crayfish, *Orconectes nais*	A	E	S	U	740	740	740	740	Sanders, 1972; Mayer and Ellersieck, 1986
Damselfly, *Ischnura verticalis*	J	E	S	U	12	12	12	12	Mayer and Ellersieck, 1986
Stonefly, *Claassenia sabulosa*	J	E,W	S	U	0.58	0.58	0.58	0.58	Mayer and Ellersieck, 1986
Stonefly, *Petronarcys californica*	J	I,E	S	U	0.5	0.5	0.5	0.5	Mayer and Ellersieck, 1986
Stonefly, *Pteronarcella badia*	J	I,E	S	U	0.5	0.5	0.5	0.5	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	9.9	—	—	—	Katz, 1961
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	9.59	—	—	—	Douglas et al., 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	2.4	—	—	—	Macek et al., 1969
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	1.1	—	—	—	Macek et al., 1969
Rainbow trout, *Oncorhynchus mykiss*	J	W	S	U	1.4	—	—	—	Macek et al., 1969
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	1.2	—	—	—	Mayer and Ellersieck, 1986

A-3

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L)				Reference
						HMAV		Overall GMAV[h]	
					Test	Species[f]	Genus[g]		
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	1.2	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	1.2	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	1.4	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	1.5	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	X	W	S	U	2.3	—	—	—	Mayer and Ellersieck, 1986
Rainbow trout, *Oncorhynchus mykiss*	J	W	FT	M	8.23	8.23	—	—	Brooke, 1993a
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	10.8	10.8	—	—	Katz, 1961
Chinook salmon, *Oncorhynchus tshawytscha*	X	W	FT	U	1.54	—	—	—	Schoettger, 1970
Chinook salmon, *Oncorhynchus tshawytscha*	J	W	S	U	6.1	3.065	—	—	Katz, 1961
Cutthroat trout, *Oncorhynchus clarki*	X	W	S	U	6.0	6.0	6.358	6.358	Mayer and Ellersieck, 1986
Goldfish, *Carassius auratus*	J	W	S	U	41	—	—	—	Henderson et al., 1959
Goldfish, *Carassius auratus*	J	W	S	U	1.6	—	—	—	Mayer and Ellersieck, 1986
Goldfish, *Carassius auratus*	J	W	S	U	1.8	4.906	4.906	4.906	Mayer and Ellersieck, 1986

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Fathead minnow, *Pimephales promelas*	J	W	S	U	18	—	—	—	Henderson et al., 1959
Fathead minnow, *Pimephales promelas*	J	W	S	U	18	—	—	—	Henderson et al., 1959
Fathead minnow, *Pimephales promelas*	J	W	S	U	36	—	—	—	Tarzwell and Henderson, 1957
Fathead minnow, *Pimephales promelas*	J	W	S	U	24	—	—	—	Tarzwell and Henderson, 1957
Fathead minnow, *Pimephales promelas*	J	W	S	U	16	—	—	—	Tarzwell and Henderson, 1957
Fathead minnow, *Pimephales promelas*	J	W	S	U	25	—	—	—	Tarzwell and Henderson, 1957
Fathead minnow, *Pimephales promelas*	J	W	S	U	23	—	—	—	Tarzwell and Henderson, 1957
Fathead minnow, *Pimephales promelas*	J	W	S	U	3.8	—	—	—	Mayer and Ellersieck, 1986
Fathead minnow, *Pimephales promelas*	L	W	FT	M	6.59	—	—	—	McCauley, 1997
Fathead minnow, *Pimephales promelas*	J	W	FT	M	4.45	5.415	5.415	5.415	McCauley, 1997
Guppy, *Poecilia reticulata*	J	W	S	U	3.616	—	—	—	Chadwick and Kiigemagi, 1968

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	Test	LC50/EC50[e] (µg/L)		Overall GMAV[h]	Reference
						HMAV			
						Species[f]	Genus[g]		
Guppy, *Poecilia reticulata*	J	W	S	U	3.219	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.912	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.306	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.328	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.496	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.047	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.430	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.047	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.672	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.645	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	6.048	—	—	—	Chadwick and Kiigemagi, 1968

Common Name, Scientific Name	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	Genus[g]	Overall GMAV[h]	Reference
Guppy, *Poecilia reticulata*	J	W	S	U	7.869	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.000	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.666	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.290	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.262	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.754	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	7.458	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	6.552	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	6.893	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	6.975	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	9.100	—	—	—	Chadwick and Kiigemagi, 1968

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L)			Overall GMAV[h]	Reference
						HMAV			
					Test	Species[f]	Genus[g]		
Guppy, *Poecilia reticulata*	J	W	S	U	5.940	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.818	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	5.865	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.713	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	6.375	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.563	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.181	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.488	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.173	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	4.032	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.569	—	—	—	Chadwick and Kiigemagi, 1968

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L)				Reference
					Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	
Guppy, *Poecilia reticulata*	J	W	S	U	3.010	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	3.280	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	J	W	S	U	2.660	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	X	W	S	U	3.431	—	—	—	Chadwick and Kiigemagi, 1968
Guppy, *Poecilia reticulata*	X	W	S	U	25	—	—	—	Henderson et al., 1959
Guppy, *Poecilia reticulata*	X	W	S	U	21	—	—	—	Cairns and Loos, 1966
Guppy, *Poecilia reticulata*	J	W	S	U	3.2	—	—	—	Adema and Vink, 1981
Guppy, *Poecilia reticulata*	J	W	S	U	7	4.313	4.313	4.313	Adema and Vink, 1981
Green sunfish, *Lepomis cyanellus*	J	W	S	U	6	—	—	—	Tarzwell and Henderson, 1957
Green sunfish, *Lepomis cyanellus*	J	W	S	U	11	—	—	—	Tarzwell and Henderson, 1957
Green sunfish, *Lepomis cyanellus*	J	W	S	U	8	8.082	—	—	Tarzwell and Henderson, 1957
Bluegill, *Lepomis macrochirus*	J	W	S	U	9	—	—	—	Henderson et al., 1959

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	17	—	—	—	Macek et al., 1969
Bluegill, *Lepomis macrochirus*	J	W	S	U	14	—	—	—	Macek et al., 1969
Bluegill, *Lepomis macrochirus*	J	W	S	U	8.8	—	—	—	Macek et al., 1969
Bluegill, *Lepomis macrochirus*	J	W	S	U	32	—	—	—	Tarzwell and Henderson, 1957
Bluegill, *Lepomis macrochirus*	J	W	S	U	18	—	—	—	Tarzwell and Henderson, 1957
Bluegill, *Lepomis macrochirus*	J	W	S	U	8	—	—	—	Tarzwell and Henderson, 1957
Bluegill, *Lepomis macrochirus*	J	W	S	U	22	—	—	—	Tarzwell and Henderson, 1957
Bluegill, *Lepomis macrochirus*	J	W	S	U	3.1	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	4.7	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	16.0	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	18.0	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	14.5	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	9.3	—	—	—	Mayer and Ellersieck, 1986

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L)				Reference
					Test	HMAV Species[f]	Genus[g]	Overall GMAV[h]	
Bluegill, *Lepomis macrochirus*	J	W	S	U	7.1	—	—	—	Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	FT	U	3.9	10.71	9.304	9.304	Mayer and Ellersieck, 1986
Largemouth bass, *Micropterus dolomieu*	X	W	S	U	3.5	3.5	3.5	3.5	Mayer and Ellersieck, 1986
Black bullhead, *Ictalurus melas*	X	E	S	U	10.0	10.0	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	X	E	S	U	4.5	—	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	X	E	S	U	7.8	5.924	7.697	7.697	Mayer and Ellersieck, 1986
Tilapia, *Tilapia mossambica*	J	W	S	U	9.2	—	—	—	Mayer and Ellersieck, 1986
Tilapia, *Tilapia mossambica*	J	W	S	U	10.0	9.592	9.592	9.592	Mayer and Ellersieck, 1986
Fowler's toad, *Bufo fowleri*	L	E	S	U	150	150	150	150	Mayer and Ellersieck, 1986
Western chorus frog, *Pseudocris triseriata*	L	E	S	U	100	100	100	100	Mayer and Ellersieck, 1986
Saltwater Species									
Polychaete worm, *Ophryotroch a diadema*	L	I	R	U	>100	—	—	—	Hooftman and Vink, 1980
Polychaete worm, *Ophryotroch a diadema*	A	I	R	U	>100	>100	>100	>100	Hooftman and Vink, 1980

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	Genus[g]	Overall GMAV[h]	Reference
Eastern oyster, *Crassostrea virginica*	E,L	W	S	U	640	640	640	640	Davis and Hidu, 1969
Mysid, *Americamysis bahia*	A	E	S	U	3.7	—	—	—	U.S. EPA, 1987b
Mysid, *Americamysis bahia*	A	E	FT	M	4.5	4.5	4.5	4.5	U.S. EPA, 1987b
Sand shrimp, *Crangon septemspinosa*	A	E	S	U	7	7	7	7	Eisler, 1969
Hermit crab, *Pagurus longicarpus*	A	E	S	U	18	18	18	18	Eisler, 1969
Grass shrimp, *Palaemonetes vulgaris*	A	E,W	S	U	50	50	—	—	Eisler, 1969
Grass shrimp, *Palaemonetes pugio*	A	E,W	FT	M	8.64	8.64	20.78	20.78	Parrish et al., 1973
Korean shrimp, *Palaemon macrodactylus*	A	E,W	S	U	16.9	—	—	—	Schoettger, 1970
Korean shrimp, *Palaemon macrodactylus*	A	E,W	FT	U	6.9	10.80	10.80	10.80	Schoettger, 1970
Pink shrimp, *Penaeus duorarum*	A	I,E	FT	M	0.70	0.70	0.70	0.70	Parrish et al., 1973
American eel, *Anguilla rostrata*	J	E	S	U	0.9	0.9	0.9	0.9	Eisler, 1970b
Chinook salmon, *Oncorhynchus tshawytscha*	J	W	FT	U	1.47	1.47	1.47	1.47	Schoettger, 1970
Atlantic silverside, *Menidia menidia*	J	W	S	U	5	5	5	5	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
Sheepshead minnow, *Cyprinodon variegatus*	A	E,W	FT	M	10.00	10.00	10.00	10.00	Parrish et al., 1973
Mummichog, *Fundulus heteroclitus*	A	E,W	S	U	5	—	—	—	Eisler, 1970a
Mummichog, *Fundulus heteroclitus*	A	E,W	S	U	16	8.944	—	—	Eisler, 1970b
Striped killifish, *Fundulus majalis*	J	E,W	S	U	4	4	5.981	5.981	Eisler, 1970b
Threespine stickleback, *Gasterosteus aculeatus*	J	E,W	S	U	15.3	—	—	—	Katz, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	E,W	S	U	13.1	14.16	14.16	14.16	Katz, 1961
Striped bass, *Morone saxatilis*	J	E	FT	U	19.7	19.7	19.7	19.7	Korn and Earnest, 1974
Shiner perch, *Cymatogaster aggregata*	J	W	S	U	3.7	—	—	—	Earnest and Benville, 1972
Shiner perch, *Cymatogaster aggregata*	J	W	FT	U	1.50	2.356	2.356	2.356	Earnest and Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	S	U	5.0	—	—	—	Earnest and Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	FT	U	2.44	3.493	3.493	3.493	Earnest and Benville, 1972
Bluehead, *Thalassoma bifasciatum*	A	W	S	U	6	6	6	6	Eisler, 1970b

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L)				Reference
					Test	HMAV Species[f]	Genus[g]	Overall GMAV[h]	
Striped mullet, *Mugil cephalus*	A	E	S	U	23	23	23	23	Eisler, 1970b
Northern puffer, Sphaeroides maculatus	A	W	S	U	34	34	34	34	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 = infaunal, 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.

Appendix B

Summary of Data from Sediment-Spiking Experiments with Dieldrin (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 ($\mu g/g$)		Interstitial Water Conc. ($\mu g/L$)	DOC ($\mu g/L$)	TOC (%)	LogK_{OC}[a]	Reference
		Dry Wt	OC					
West Bearskin, MN *Hyalella azteca*	30	1.43	48.15	14.9	—	2.97	—	Hoke et al., 1995
	28	3.75	148.81	42.3	—	2.52	—	
	38	12.34	474.62	53.9	—	2.60	—	
	45	30.69	999.67	210.0	—	3.07	—	
	63	48.73	1450.30	245.5	—	3.36	—	
Pequaywan, MN *Hyalella azteca*	13	14.48	193.58	58.8	—	7.48	—	Hoke et al., 1995
	10	43.81	541.53	146.6	—	8.09	—	
	—	123.50	1848.80	343.8	—	6.68	—	
	40	249.30	2280.88	566.1	—	10.93	—	
	55	479.37	4672.22	518.7	—	10.26	—	
Airport Pond, MN *Hyalella azteca*	15	5.17	304.1	16.70	—	1.70	—	Hoke et al., 1995
	60	25.24	1493.5	80.12	—	1.69	—	
	100	97.38	5527.3	89.40	—	1.76	—	
Airport Pond, MN *Hyalella azteca*	63	3.77	249.67	13.5	69.69	1.51	4.82	Hoke and Ankley, 1992
	30	16.71	960.34	60.3	73.46	1.74	4.76	
	27	31.55	1889.22	136.0	59.89	1.67	4.78	
	37	61.10	3432.58	224.4	63.73	1.78	4.75	
	53	136.02	7556.11	356.8	67.17	1.80	4.74	
Airport Pond, MN *Hyalella azteca*	30	7.29	402.76	30.1	66.0	1.81	4.74	Hoke and Ankley, 1992
	40	30.52	1623.4	143.3	75.5	1.88	4.73	
	47	115.78	6432.2	311.40	65.6	1.80	4.74	
Airport Pond, MN *Chironomus tentans*	5	0.09	4.95	—	—	1.82	—	Hoke et al., 1995
	55	1.00	49.26	—	—	2.03	—	
	50	5.41	252.80	—	—	2.14	—	
	90	12.98	658.88	—	—	1.97	—	
Airport Pond, MN *Chironomus tentans*	0	0.05	3.45	—	—	1.45	—	Hoke et al., 1995
	5	0.10	5.92	—	—	1.69	—	
	50	0.52	36.62	—	—	1.42	—	
	100	3.78	252.00	—	—	1.50	—	
	100	9.64	614.01	—	—	1.57	—	

[a]K_{OC} (L/kg) = sediment concentration (μg_{OC}) ÷ calculated free interstitial water concentration ($\mu g/L$) × 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: Dieldrin**

All data were obtained either from the WQC document for dieldrin (USEPA, 1980) or from a comprehensive literature search completed in 1997.

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* (U.S. EPA, 2000).

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