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EPA

Equilibrium Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms: Dieldrin

DRAFT

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 meet the objectives of the CWA, EPA has periodically issued ambient water quality criteria (WQC) beginning with the publication of "Water Quality Criteria, 1972" (NAS, 1973). The development of WQC is authorized by Section 304(a)(1) of the CWA, which directs the Administrator to develop and publish "criteria" reflecting the latest scientific knowledge on (1) the kind and extent of effects on human health and welfare, including effects on plankton, fish, shellfish, and wildlife, that may be expected from the presence of pollutants in any body of water, including ground water; and (2) the concentration and dispersal of pollutants on biological community diversity, productivity, and stability. All criteria guidance through late 1986 was summarized in an EPA document entitled "Quality Criteria for Water, 1986" (U.S. EPA, 1987a). Updates on WQC documents for selected chemicals and new criteria recommendations for other pollutants have been more recently published as "National Recommended Water Quality Criteria- Correction" (U.S. EPA, 1999). EPA will continue to update the nationally recommended WQC as needed in the future.

In addition to the development of WQC and to continue to meet the objectives of the CWA, EPA has conducted efforts to develop and publish equilibrium partitioning sediment guidelines (ESGs) 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. These guidelines are authorized under Section 304(a)(2) of the CWA, which directs the Administrator to develop and publish information on, among other things, the factors necessary to restore and maintain the chemical, physical, and biological integrity of all navigable waters.

The ESGs and associated methodology presented in this document are EPA's best recommendation as to the concentrations of a substance that may be present in sediment while still protecting benthic organisms from the effects of that substance. These guidelines 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 ESGs 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 ESGs do not protect against synergistic or antagonistic effects of contaminants or bioaccumulative effects to benthos. They are not protective of wildlife or human health endpoints.

EPA recommends that ESGs be used 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. EPA is developing guidance to assist in the application of these guidelines in water-related programs of the States and this Agency. This document provides guidance to EPA Regions, States, the regulated community, and the public. It is designed to implement national policy concerning the matters addressed. It does not, however, 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 guidance where appropriate. EPA may change this guidance in the future.

Foreword

This document has been reviewed by EPA's Office of Science and Technology (Health and Ecological Criteria Division, Washington, DC) and Office of Research and Development (Mid-Continent Ecology Division, Duluth, MN, Atlantic Ecology Division, Narragansett, RI), and approved for publication.

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Executive Summary

This equilibrium partitioning sediment guideline (ESG) document recommends a sediment concentration for the insecticide dieldrin that is EPA's best estimate of the concentration 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 incorporation of the appropriate biological effects concentration. This provides for the derivation of a guideline that is causally linked to the specific chemical, applicable across sediments, and appropriately protective of benthic organisms.

EqP theory asserts 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 concentration 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 ESG Technical Basis Document (U.S EPA, 2000a) 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}. Because the water quality criteria (WQC) represent the concentration of a chemical in water that is protective of the presence of aquatic life, and is appropriate for benthic organisms, the product of the final chronic value (FCV) from the WQC and K_{OC} is the concentration in sediments that, on an organic carbon basis, is protective of 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 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 ESG should be interpreted as a chemical concentration below which adverse effects are not expected. In comparison, at concentrations above the ESG effects are likely, and 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 ESG and associated uncertainty limits.

These guidelines do not protect against additive, synergistic, or antagonistic effects of contaminants or bioaccumulative effects to aquatic life, wildlife, or human health. The Agency and the EPA Science Advisory Board do not recommend the use of ESGs as stand-alone, pass-fail criteria for all applications; rather, ESGs could trigger additional studies at sites under investigation. This ESG applies only to sediments having ≥0.2% organic carbon.

EPA has developed both Tier 1 and Tier 2 ESGs to reflect the differing degrees of data availability and uncertainty. Requirements for a Tier 1 ESG include a K_{OW}, FCV, and sediment toxicity tests to verify EqP assumptions. In comparison, a Tier 2 ESG requires a K_{OW} and a FCV or secondary chronic value (SCV); sediment toxicity tests are recommended but not required. The ESGs derived for dieldrin in this document, as well as the ESGs for endrin, metal mixtures (Cd, Cu, Pb, Ni, Ag, Zn), and polycyclic aromatic hydrocarbon (PAH) mixtures represent Tier 1 ESGs (U.S. EPA, 2000d,e,f). Information on how EPA recommends ESGs be applied in specific regulatory programs is described in the "Implementation Framework for the Use of Equilibrium Partitioning Sediment Guidelines (ESGs)" (EPA, 2000c).

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
ESG(s)	Equilibrium partitioning sediment guideline(s); for nonionic organics, this term usually refers to a value that is organic carbon–normalized (more formally ESG_{OC}) unless otherwise specified
$ESG_{dry\ wt}$	Dry weight–normalized equilibrium partitioning sediment guideline
ESG_{OC}	Organic carbon–normalized equilibrium partitioning sediment guideline
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
OBC	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
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

Under the Clean Water Act (CWA) the U.S. Environmental Protection Agency (EPA) is responsible for protecting the chemical, physical, and biological integrity of the nation's waters. In keeping with this responsibility, EPA published ambient water quality criteria (WQC) in 1980 for 64 of the 65 toxic pollutants or pollutant categories designated as toxic in the CWA. Additional water quality documents that update criteria for selected consent decree chemicals and new criteria have been published since 1980. These WQC are numerical concentration limits that are EPA's best estimate of concentrations protective of human health and the presence and uses of aquatic life. Although these WQC play an important role in ensuring a healthy aquatic environment, they alone are not sufficient to ensure the protection of environmental or human health.

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 guidelines 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 guidelines derivation in all situations (U.S. EPA, 1989a). The equilibrium partitioning (EqP) approach was selected for nonionic organic chemicals because it presented the greatest promise for generating defensible, national, numerical chemical-specific guidelines applicable across a broad range of sediment types. The three principal observations that underlie the EqP approach of establishing sediment guidelines 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 final chronic values (FCV) 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 equilibrium partitioning sediment guideline (ESG), derived as the product of the K_{OC} and FCV, is protective of benthic organisms. ESG concentrations presented in this document are expressed as μg chemical/g sediment

organic carbon ($\mu g/g_{OC}$) and not on an interstitial water basis because (1) interstitial water is difficult to sample and (2) significant amounts of the dissolved chemical may be associated with dissolved organic carbon; thus, total chemical concentrations in interstitial water may overestimate exposure.

Sediment guidelines generated using the EqP approach (i.e., ESGs) are suitable for use in providing guidance 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

ESGs 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, ESGs 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. ESGs 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 ESGs under these conditions may result in values lower or higher than those presented in this document.

The ESG values presented herein represent EPA's best recommendation of the concentration of dieldrin in sediment that will not adversely affect most benthic organisms. EPA recognizes that these ESG 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 the ESG 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 the ESG 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, the national ESG would be overly protective of benthic organisms and should not be used unless modified using the procedures outlined in "Methods for the Derivation of Site-Specific Equilibrium Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms" (U.S EPA, 2000b). The ESG may be underprotective where the toxicity of other chemicals are additive with the ESG 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 ESG for dieldrin. The data that support the EqP approach for deriving an ESG for nonionic organic chemicals are reviewed by Di Toro et al. (1991) and EPA (U.S. EPA, 2000a). Before proceeding through the following text, tables, and calculations, the reader should consider reviewing "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (Stephan et al., 1985), "Response to Public Comment" (U.S. EPA, 1985), and "Technical Basis for the Derivation of Equilibrium Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms: Nonionic Organics (U.S. EPA, 2000a). Guidance for acceptable use of ESG values is contained in "Implementation Framework for the Use of Equilibrium Partitioning Sediment Guidelines (ESGs)" (U.S. EPA, 2000c).

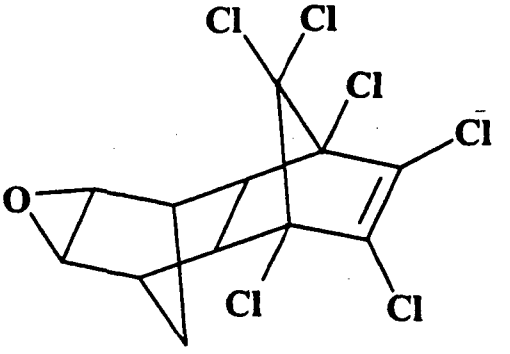
1.2 General Information: Dieldrin

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 endrin, 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°C and specific gravity of 1 75 g/cc at 20°C. It has a vapor pressure of 0.4 mPa (20°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



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).

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 ESG for dieldrin is effects based. It is calculated from the FCV derived in Section 3.

1.3 Applications of Sediment Guidelines

ESGs are meant to be used with direct toxicity testing of sediments as a method of evaluation. 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 endrin, metal mixtures (Cd, Cu, Pb, Ni, Ag, Zn), and polycyclic aromatic hydrocarbon (PAH) mixtures.

EPA has developed both Tier 1 and Tier 2 ESGs to reflect the differing degrees of data availability and uncertainty. The minimum requirements to derive a Tier 1 ESG include (1) an octanol–water partitioning coefficient (K_{ow}) of the chemical, measured with current experimental techniques, which appears to remove the large variation in reported values; (2) derivation of the FCV, which should also be updated to include the most recent toxicological information; and (3) sediment toxicity "check" tests to verify EqP predictions. Check experiments can be used to verify the utility of EqP for a particular chemical. As such, the ESGs derived for nonionic organics, such as dieldrin and endrin, metal mixtures, and PAH mixtures represent Tier 1 ESGs (U.S. EPA, 2000d,e,f). In comparison, the minimum requirements for a Tier 2 ESG include a K_{ow} for the chemical (as described above) and the use of either a FCV or secondary chronic value (SCV). The performance of sediment toxicity tests is recommended, but not required for the development of Tier 2 ESGs. Therefore, in comparison to Tier 1 ESGs, the level of protection provided by the Tier 2 ESGs would be associated with more uncertainty due to the use of the SCV and absence of sediment toxicity tests. Examples of Tier 2 ESGs for nonionics are found in U.S. EPA (2000g). Information on how EPA recommends ESGs be applied in specific regulatory programs is described in the "Implementation Framework for the Use of Equilibrium Partitioning Sediment Guidelines (ESGs)" (EPA, 2000c).

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 ESG. 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 ESG concentration. 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 ESG. 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 ESG for dieldrin and its uncertainty. The ESG for dieldrin is then compared with three databases on dieldrin's environmental occurrence in sediments. Section 6 concludes with the guideline statement for dieldrin. The references cited in this document are listed in Section 7.

Section 2

Partitioning

2.1 Description of EqP Methodology

ESGs 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. As a result, they can be used in much the same way as WQC; that is, the concentration of a chemical that is protective of the intended use, such as aquatic life protection. For nonionic organic chemicals, ESGs are expressed as μg chemical/g_{OC} and apply to sediments having ≥0.2% organic carbon by dry weight. A brief overview follows of the concepts that underlie the EqP methodology for deriving ESGs. The methodology is discussed in detail in "Technical Basis for the Derivation of Equilibrium Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms: Nonionic Organics" (U.S. EPA, 2000a), hereafter referred to as the ESG 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 for determining ESGs 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 ESGs. For this reason, the methodology described below is called the EqP method

The ESG 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 ESG 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 the ESG.

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

$$ESG = K_P FCV \quad (2\text{-}1)$$

This is the fundamental equation used to generate the ESG. 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

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

It follows that

$$ESG_{OC} = K_{OC} FCV \quad (2\text{-}3)$$

where ESG_{OC} is the ESG on a sediment organic carbon basis. For nonionic organics, the ESG term usually refers to a value that is organic carbon–normalized (more formally ESG_{OC}) 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 ESG, as discussed in the ESG 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) will be used to derive the ESG 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

$$K_p = \frac{f_{OC}K_{OC}}{1 + mf_{OC}K_{OC}/\upsilon_X} \qquad (2\text{-}4)$$

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

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

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

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

Method	$Log_{10}K_{OW}$	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

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 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

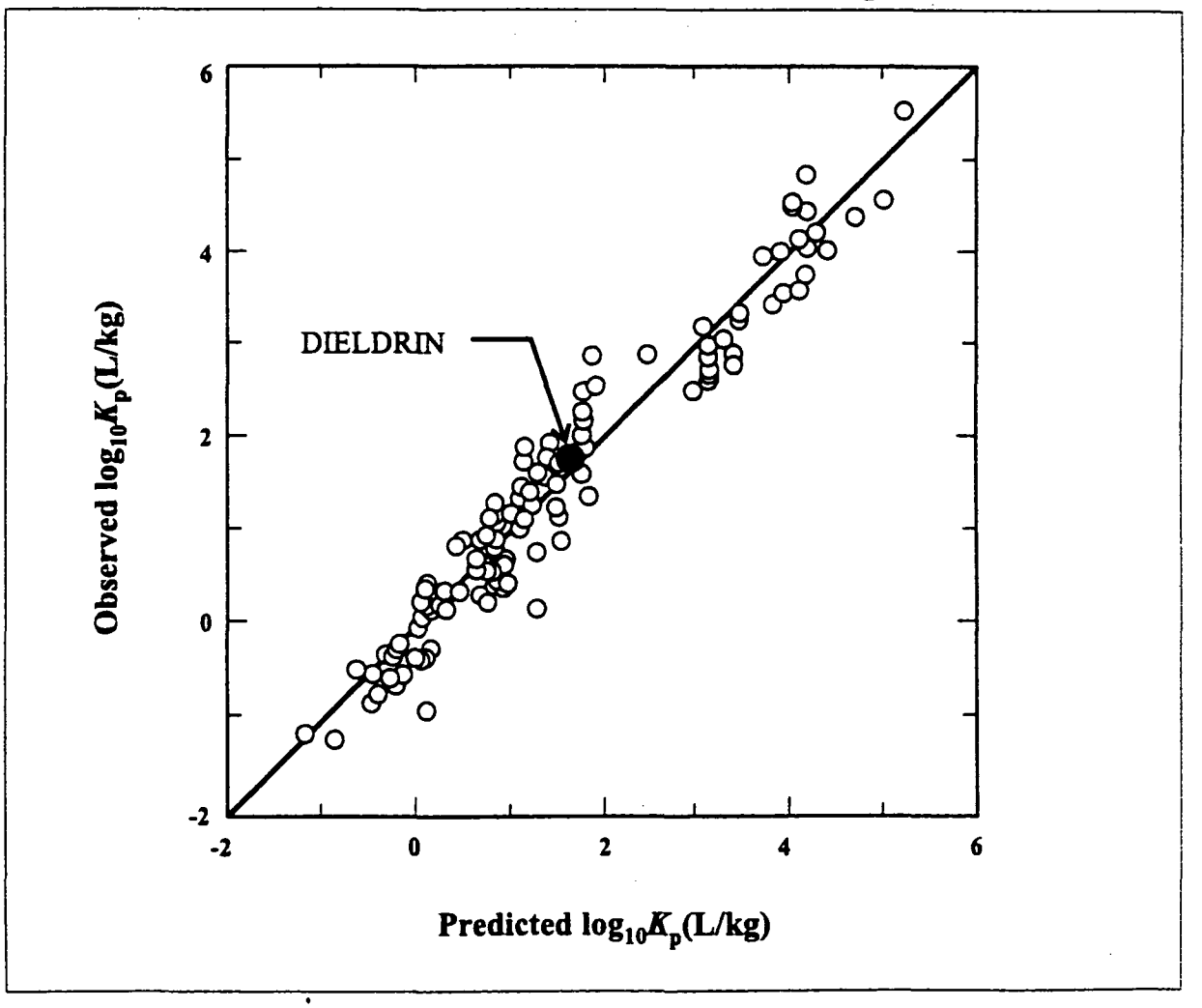


Figure 2-1. Observed versus predicted partition coefficients for nonionic organic chemicals, using Equation 2-4 (figure from Di Toro, 1985). Dieldrin datum is highlighted (Sharom et al., 1980).

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

Observed $Log_{10}K_{OC}$ (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

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 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 ($\mu g/g_{OC}$) is plotted versus the calculated free (dissolved) interstitial water concentration ($\mu g/L$). 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 ($\mu g/L$) presented in Figure 2-2 is given by

$$C_d = \frac{C_{TW}}{1 + m_{DOC}K_{DOC}} \qquad (2\text{-}6)$$

where C_{TW} 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 $\log_{10}K_{OC} = 5.28$, derived from the dieldrin $\log_{10}K_{OW}$ 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 $\log_{10}K_{OC} = 5.28$ (Figure 2-2A).

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

2.4 Summary of Derivation of K_{OC} for Dieldrin

The K_{OC} selected to calculate the ESG for dieldrin was based on the regression of $\log_{10}K_{OC}$ to $\log_{10}K_{OW}$ (Equation 2-5) using the dieldrin $\log_{10}K_{OW}$ 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 $\log_{10}K_{OC} = 5.28$. This value was in agreement with the $\log_{10}K_{OC}$ of 5.32 measured in the sediment toxicity tests.

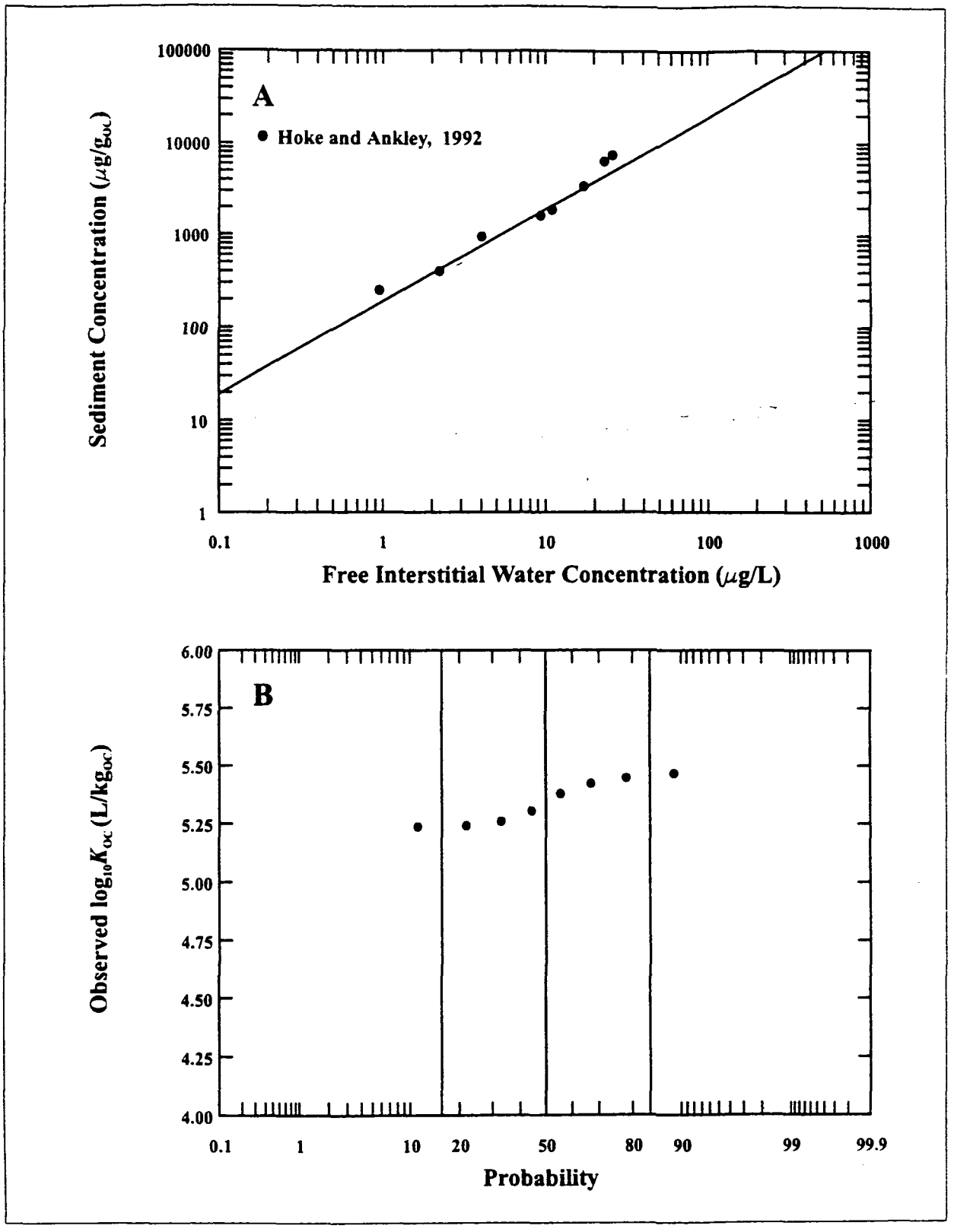


Figure 2-2. Organic carbon-normalized sorption isotherm for dieldrin (A) and probability plot of K_{OC} (B) from sediment toxicity tests (Hoke and Ankley, 1992). The solid line represents the relationship predicted with a $\log_{10}K_{OC}$ of 5.28.

Section 3

Toxicity of Dieldrin in Water Exposures

3.1 Derivation of Dieldrin WQC

The EqP method for derivation of the ESG 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 effects concentration for the ESG derivation.

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 among both 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

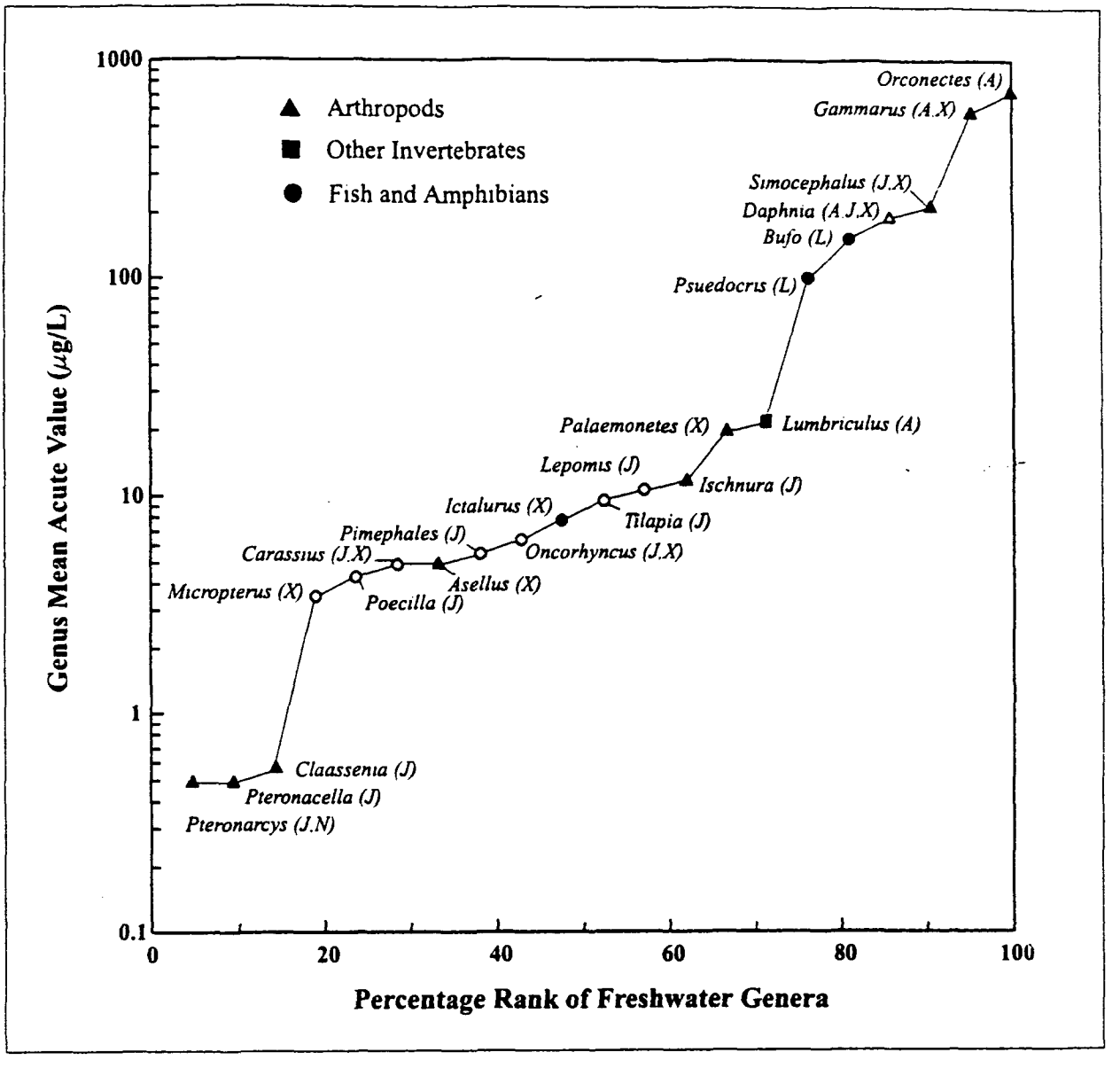


Figure 3-1. Genus mean acute values from water-only acute toxicity tests using freshwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. A=adult, J=juvenile, N=naiads, X=unspecified life-stage.

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

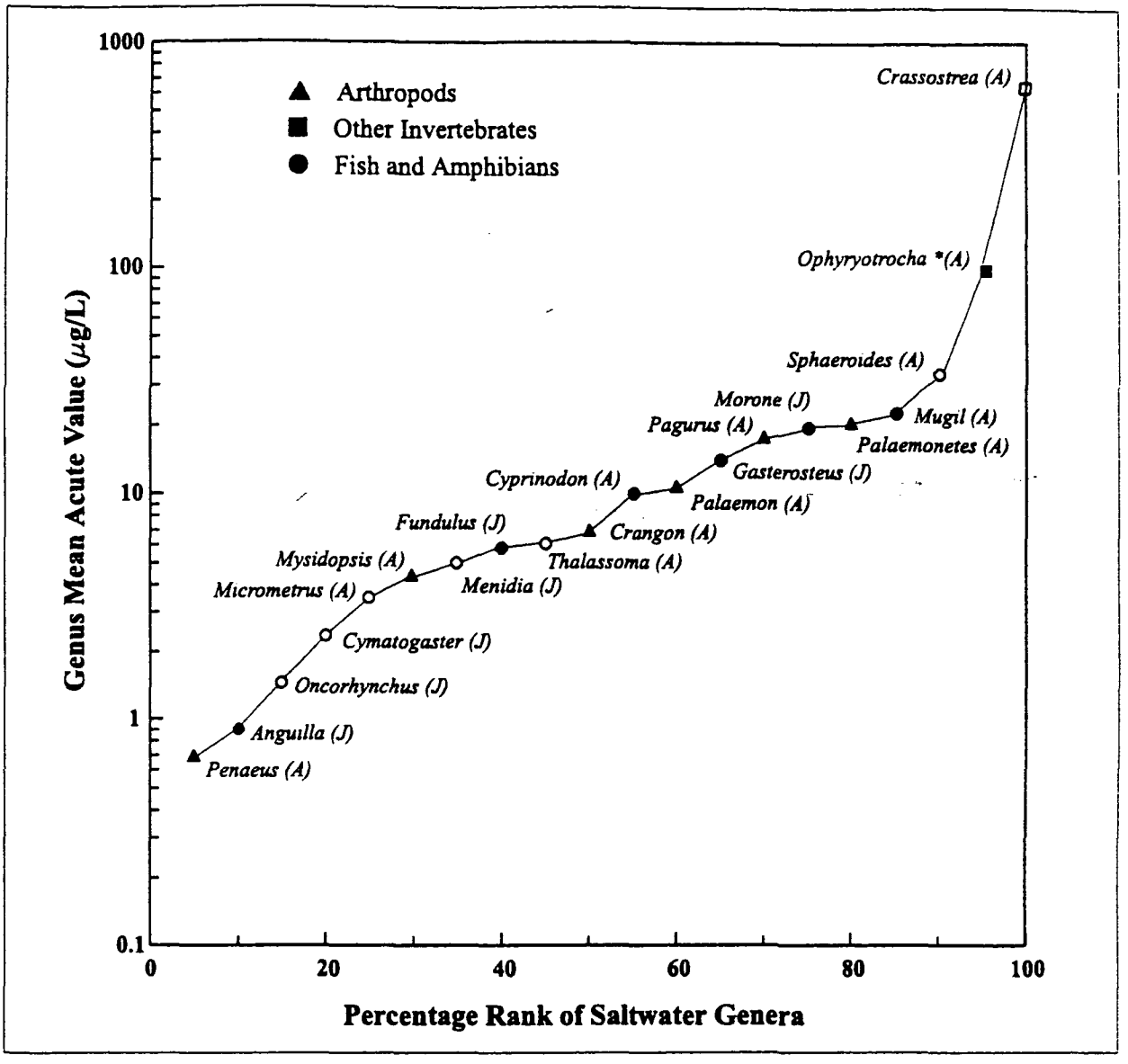


Figure 3-2. Genus mean acute values from water-only acute toxicity tests using saltwater species versus percentage rank of their sensitivity. Symbols representing benthic species are solid; those representing water column species are open. Asterisk indicates greater than values. A=adult, J=juvenile.

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)

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 ESG 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 the Dieldrin ESG

Use of the FCV as the effects concentration for calculation of the ESG 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 ESG Technical Basis Document (U.S. EPA, 2000a) 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, 2000a). 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.

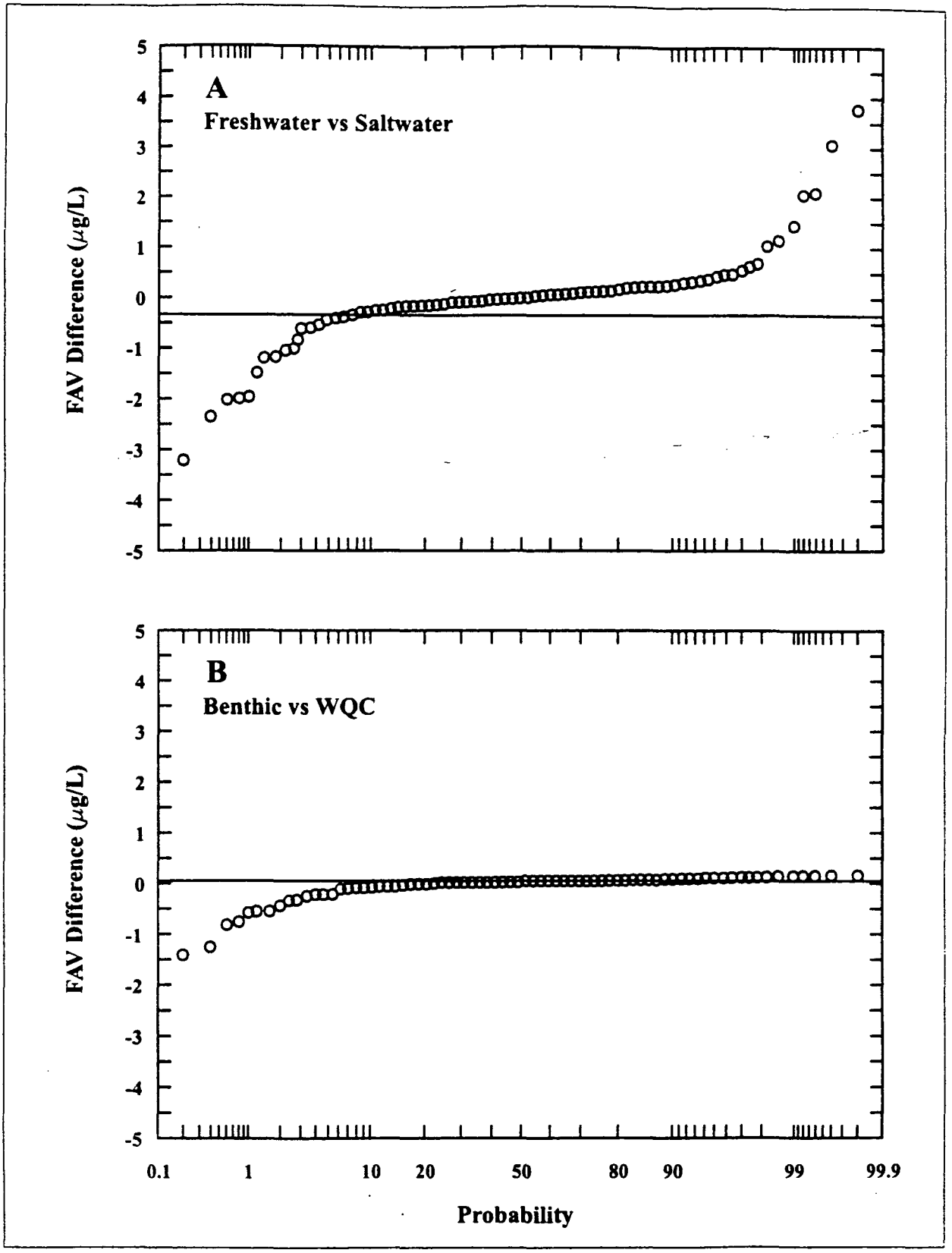


Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater versus saltwater (A) and benthic versus WQC (B) data. The solid lines in the figure correspond to the FAV differences measured for dieldrin.

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 $\mu g/g_{OC}$).

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 $\mu g/g_{OC}$). 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 $\mu g/g_{OC}$). Concentrations of dieldrin in water overlying 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)	Sediment Dieldrin LC50 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.

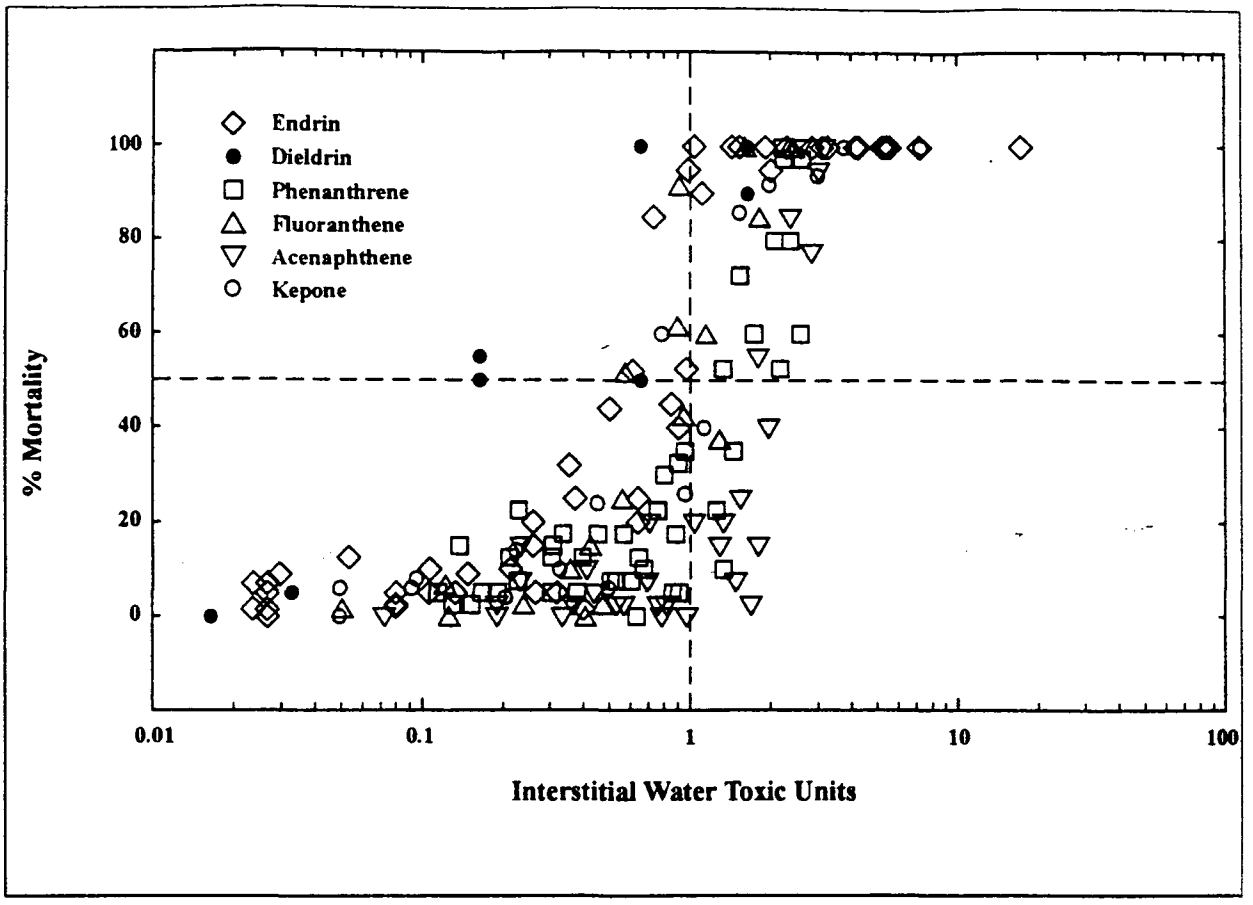


Figure 4-1. Percent mortalities of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and midge in sediments spiked with dieldrin (Hoke et al., 1995) or kepone (Adams et al., 1985) relative to interstitial water units.

sediment were 10 times the LC50 in water. The authors concluded that sediment-associated dieldrin contributed little to the toxicity observed.

The need for organic carbon normalization of the concentration of nonionic organic chemicals in sediments is presented in the ESG 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 guidelines 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, 2000a). 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

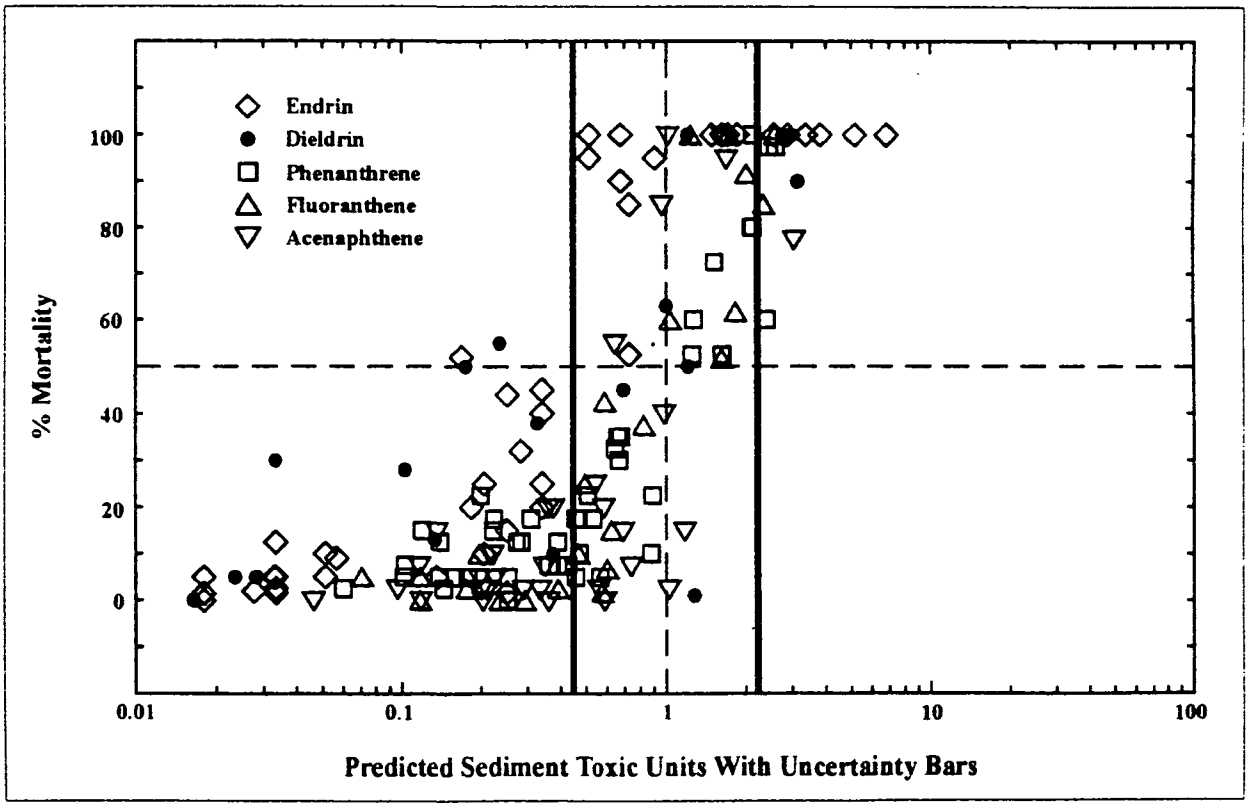


Figure 4-2. **Percent mortalities of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke et al., 1995), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992), and midge in sediments spiked with dieldrin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

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). 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 ESG 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.

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: Dry Wt. (μg/g)	Dieldrin Sediment LC50s: OC (μg/g)	Predicted[b] LC50 ($\mu g/g_{OC}$)	Ratio: Actual/ Predicted LC50	Reference
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 ($\mu g/g_{OC}$) = 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.

4.3 Tests of the Equilibrium Partitioning Prediction of Sediment Toxicity

Sediment guidelines derived using the EqP approach utilize partition coefficients and FCVs from updated or final WQC documents to derive the ESG concentration that is 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 ESG 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 $\pm 1.96 \times$ (ESG uncertainty). The value of 1.96 is the t statistic, which provides a 95% confidence interval around the ESG.

Section 5

Guidelines Derivation for Dieldrin

5.1 Guidelines Derivation

The WQC FCV (see Section 3), without an averaging period or return frequency, is used to calculate the ESG 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 ESG, 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 WQC Guidelines (Stephan et al., 1985).

The EqP method for calculating ESGs 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 ESG (μg/g sediment) is computed using the partition coefficient, K_P (L/g sediment), between sediment and interstitial water

$$ESG = K_P FCV \quad (5\text{-}1)$$

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

$$ESG_{OC} = K_{OC} FCV \quad (5\text{-}2)$$

Because K_{OC} is presumably independent of sediment type for nonionic organic chemicals, so too is ESG_{OC}. Table 5-1 contains the calculation of the dieldrin ESG.

Table 5-1. Equilibrium partitioning sediment guidelines (ESGs) for dieldrin

Type of Water Body	$Log_{10}K_{OW}$ (L/kg)	$Log_{10}K_{OC}$ (L/kg)	FCV (μg/L)	ESG_{OC} ($\mu g/g_{OC}$)
Freshwater	5.37	5.28	0.06589	12[a]
Saltwater	5.37	5.28	0.1469	28[b]

[a] $ESG_{OC} = (10^{5.28}\ L/kg_{OC}) \times (10^{3}\ kg_{OC}/g_{OC}) \times (0.06589\ \mu g\ dieldrin/L) = 12\ \mu g\ dieldrin/g_{OC}$
[b] $ESG_{OC} = (10^{5.28}\ L/kg_{OC}) \times (10^{3}\ kg_{OC}/g_{OC}) \times (0.1469\ \mu g\ dieldrin/L) = 28\ \mu g\ dieldrin/g_{OC}$

The ESG_{OC} is applicable to sediments with f_{OC} ≥0.2%. For sediments with f_{OC} <0.2%, organic carbon normalization and ESGs do not apply.

Because organic carbon is the factor controlling the bioavailability of nonionic organic compounds in sediments, ESGs 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 ESG value. This facilitates comparisons between the ESG and field concentrations relative to identification of hot spots and the degree to which sediment concentrations do or do not exceed the ESG values. The conversion from dry weight to organic carbon–normalized concentration can be done using the following formula

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

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 $\mu g/g_{OC}$ (= 0.1 $\mu g/g_{dry\ wt} \times 100 \div 0.5$), which exceeds the freshwater dieldrin ESG of 12 $\mu g/g_{OC}$. Another freshwater sediment with the same concentration of dieldrin (0.1 $\mu g/g_{dry\ wt}$) but a TOC concentration of 5.0% would have an organic carbon–normalized concentration of 2.0 $\mu g/g_{OC}$ (= 0.1 $\mu g/g_{dry\ wt} \times 100 \div$ 5.0), which is below the freshwater ESG 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 ESG_{OC} may be "converted" to dry weight–normalized ESG values ($ESG_{dry\ wt}$). This "conversion" for each level of TOC is

$$ESG_{dry\ wt} = ESG_{OC}\,(\mu g/g_{OC}) \times (\%\ TOC \div 100)$$

For example, the $ESG_{dry\ wt}$ value for freshwater sediments with 1% organic carbon is 0.12 $\mu g/g$

$$ESG_{dry\ wt} = 12\ \mu g/g_{OC} \times 1\%\ TOC \div 100 = 0.12\ \mu g/g_{dry\ wt}$$

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

5.2 Uncertainty Analysis

Some of the uncertainty of the dieldrin ESG 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 guidelines. 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 ($\mu g/g_{OC}$) can be estimated from the product of the effects concentration from water-only exposures, FCV ($\mu g/L$), and the partition coefficient, K_{OC} (L/kg). The uncertainty associated with the ESG 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 ESG (see Section 4.3 and the ESG 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 ($LC50_W$; $\mu g/L$) are related to the organic carbon–normalized LC50 values from sediment exposures ($LC50_{S\,OC}$, $\mu g/g_{OC}$) via the partitioning equation

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

As mentioned above, one of the assertions of the EqP model is that the toxicity of sediments expressed on an organic carbon basis equals the toxicity in water-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

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

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, 2000a). The results are shown in Table 5-2 The last line of Table 5-2 is the uncertainty associated with the ESG; i.e., the variance associated with the exposure media variability.

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

$$\ln(ESG_{OC})_{UPPER} = \ln(ESG_{OC}) + 1.96\sigma_{ESG} \quad (5\text{-}5)$$

$$\ln(ESG_{OC})_{LOWER} = \ln(ESG_{OC}) - 1.96\sigma_{ESG} \quad (5\text{-}6)$$

The confidence limits are given in Table 5-3.

The ESG_{OC} is applicable to sediments with $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic carbon normalization and ESGs do not apply.

Table 5-2. Analysis of variance for derivation of confidence limits of the ESGs 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] $\sigma_{ESG} = \sigma_\alpha$

Table 5-3. Confidence limits of the ESGs 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 ESG 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 ESG 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

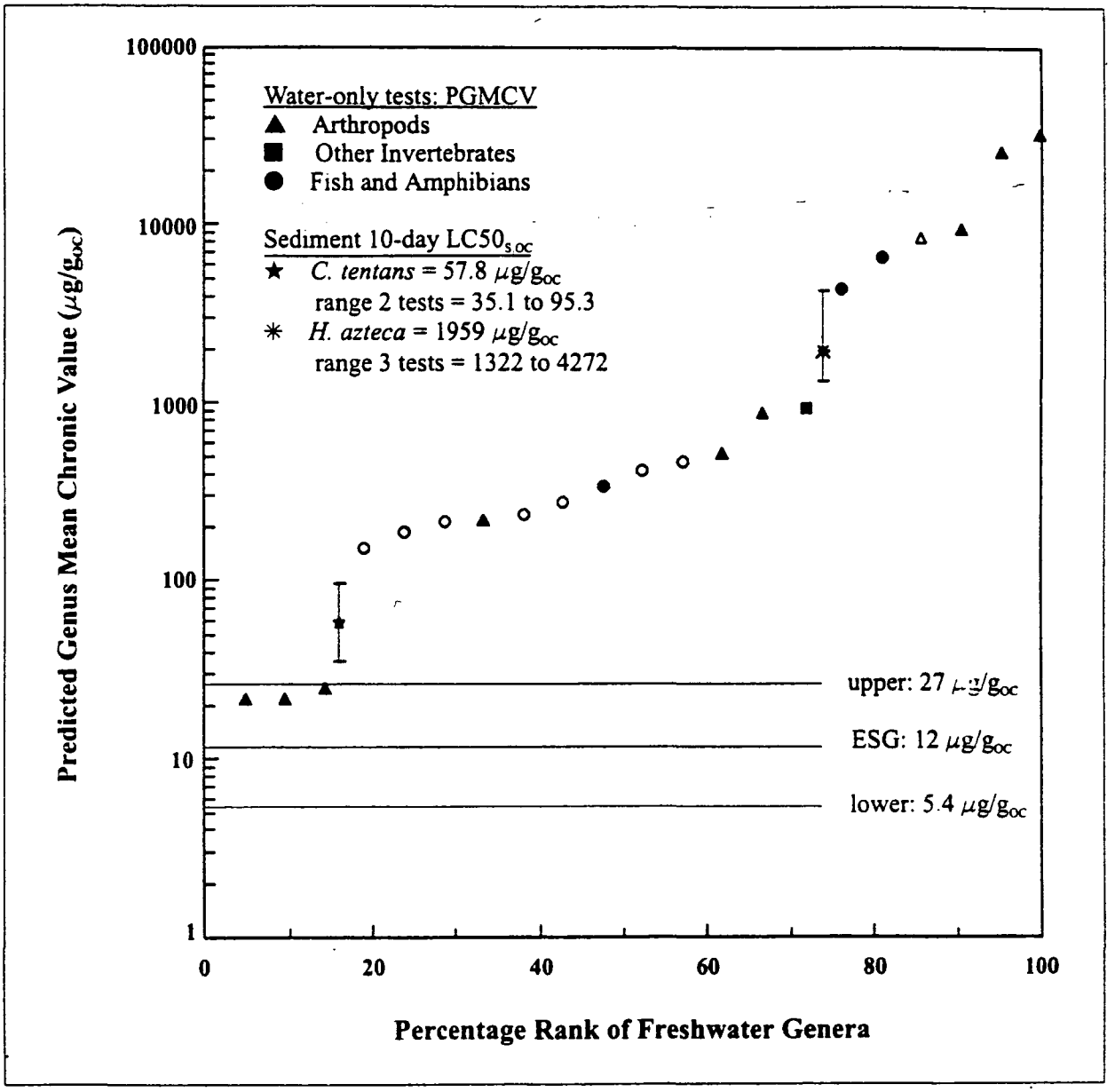


Figure 5-1. Predicted genus mean chronic values calculated from water-only toxicity values (Equation 5-7; Appendix A) using freshwater species versus percentage rank of their sensitivity. Lines indicate the freshwater dieldrin ESG ± 95% confidence limits. Solid symbols are benthic genera; open symbols 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* (★) 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 ESGs, organism sensitivities, and PGMCVs is not intended to provide accurate predictions of the responses of taxa or communities of

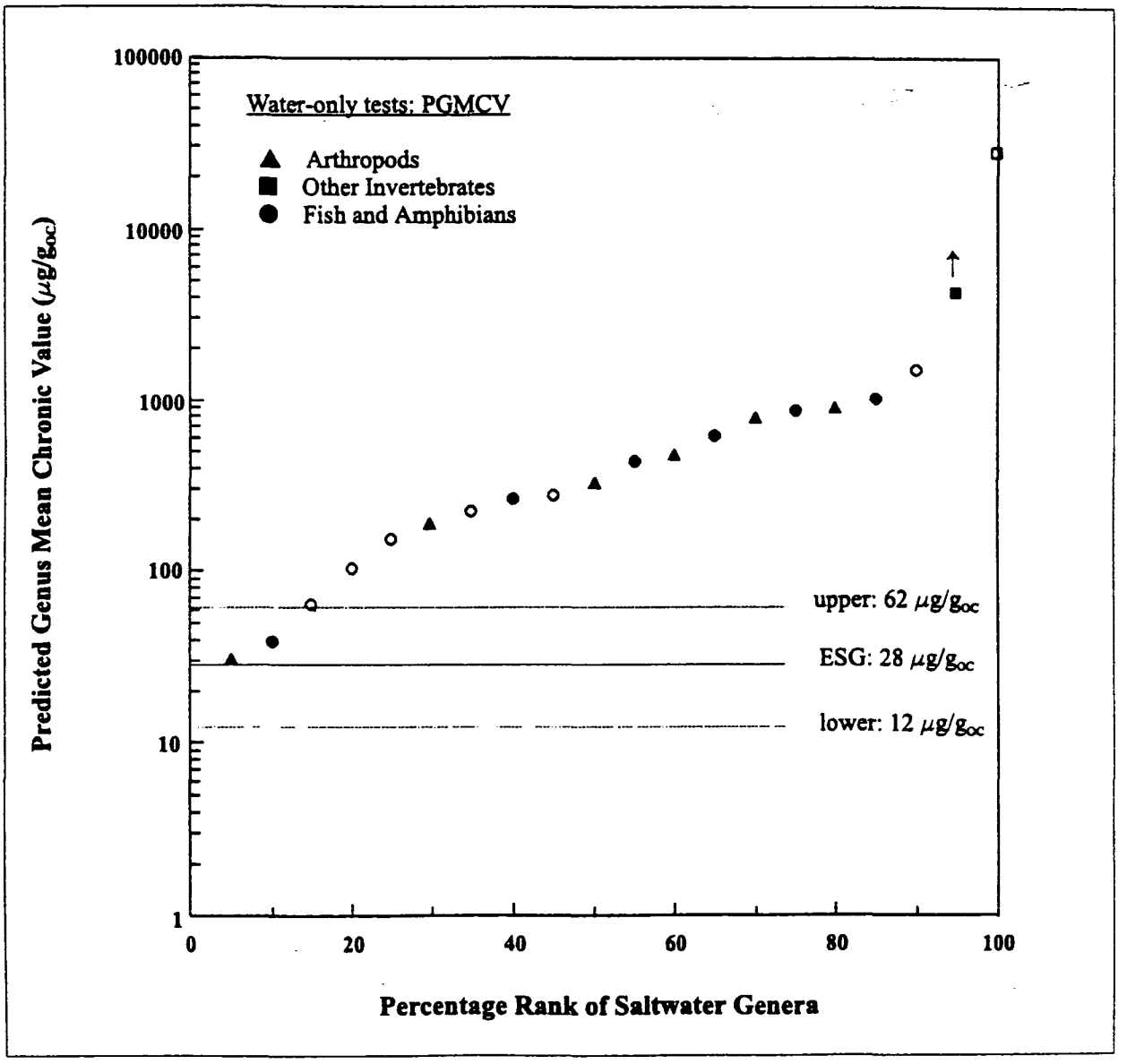


Figure 5-2. **Predicted genus mean chronic values calculated from water-only toxicity values (Equation 5-7; Appendix A) using saltwater species versus percentage rank of their sensitivity. Solid symbols are benthic genera; open symbols are water column genera. Arrows indicate greater than values.**

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 ESG for dieldrin (12 $\mu g/g_{OC}$) 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 ESG (27 $\mu g/g_{OC}$). The PGMCVs for the stoneflies *Pteronarcella* (22 $\mu g/g_{OC}$), *Pteronarcys* (22 $\mu g/g_{OC}$), and *Claassenia* (26 $\mu g/g_{OC}$) are below the ESG 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 ESG 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 ESG 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 ESG (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 ESG of 12 $\mu g/g_{OC}$, the 10-day LC50s from two tests with the midge *C. tentans* ranged from 2.9 to 7.9 times the ESG (see insert Figure 5-1; corresponding values from Table 4-1).

The saltwater ESG for dieldrin (28 $\mu g/g_{OC}$) is less than all of the PGMCVs for saltwater genera (Figure 5-2). The PGMCVs for the penaeid shrimp *Penaeus duorarum* (31 $\mu g/g_{OC}$) and the fish *Anguilla rostrata* (39 $\mu g/g_{OC}$) are lower than the upper 95% confidence interval for the ESG (62 $\mu g/g_{OC}$). 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 ESG 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 ESG.

5.4 Comparison of Dieldrin ESG 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 guidelines. In order to investigate this possibility, the dieldrin guidelines 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 ESG values were actually exceeded may be less than the percentage reported. Very few of the measured values from either of the databases exceed the ESGs.

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

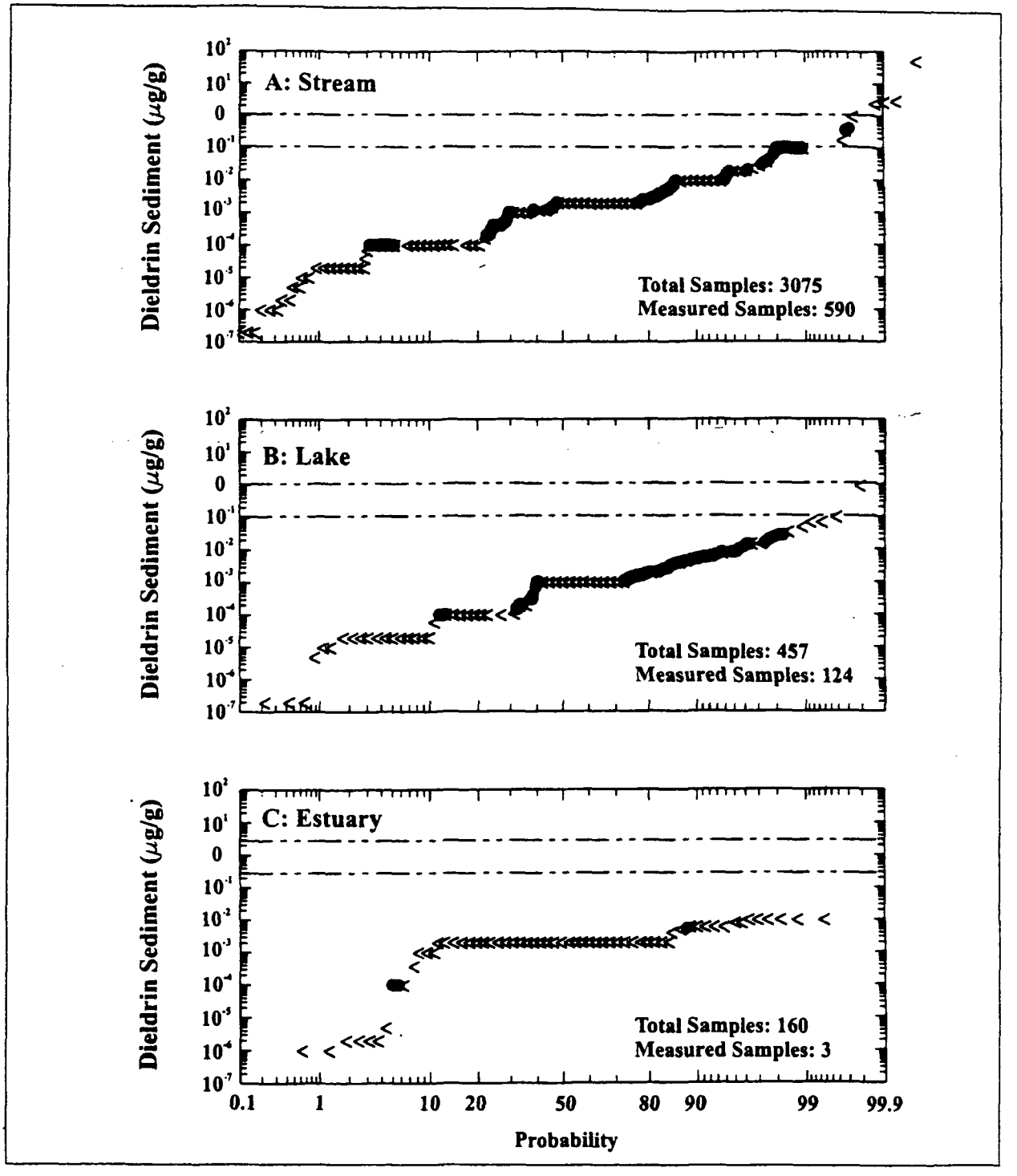


Figure 5-3. Probability distribution of concentrations of dieldrin in sediments from streams (A), lakes (B), and estuaries (C) in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared with the dieldrin ESG values. Sediment dieldrin concentrations less than the detection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

The ESG 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, ESG values corresponding to sediment organic carbon levels of 1% to 10% were compared with dieldrin's distribution in sediments as examples only. For freshwater sediments, ESG values 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, ESGs were 0.28 μg/g dry weight and 2.8 μg/g dry weight, respectively. Figure 5-3 presents comparisons of these ESGs 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 ESGs 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 guideline, 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 ESG value for 1% organic carbon sediments and three orders of magnitude below the ESG 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 ESG for dieldrin (28 $\mu g/g_{OC}$) in Figure 5-4. Data presented were from

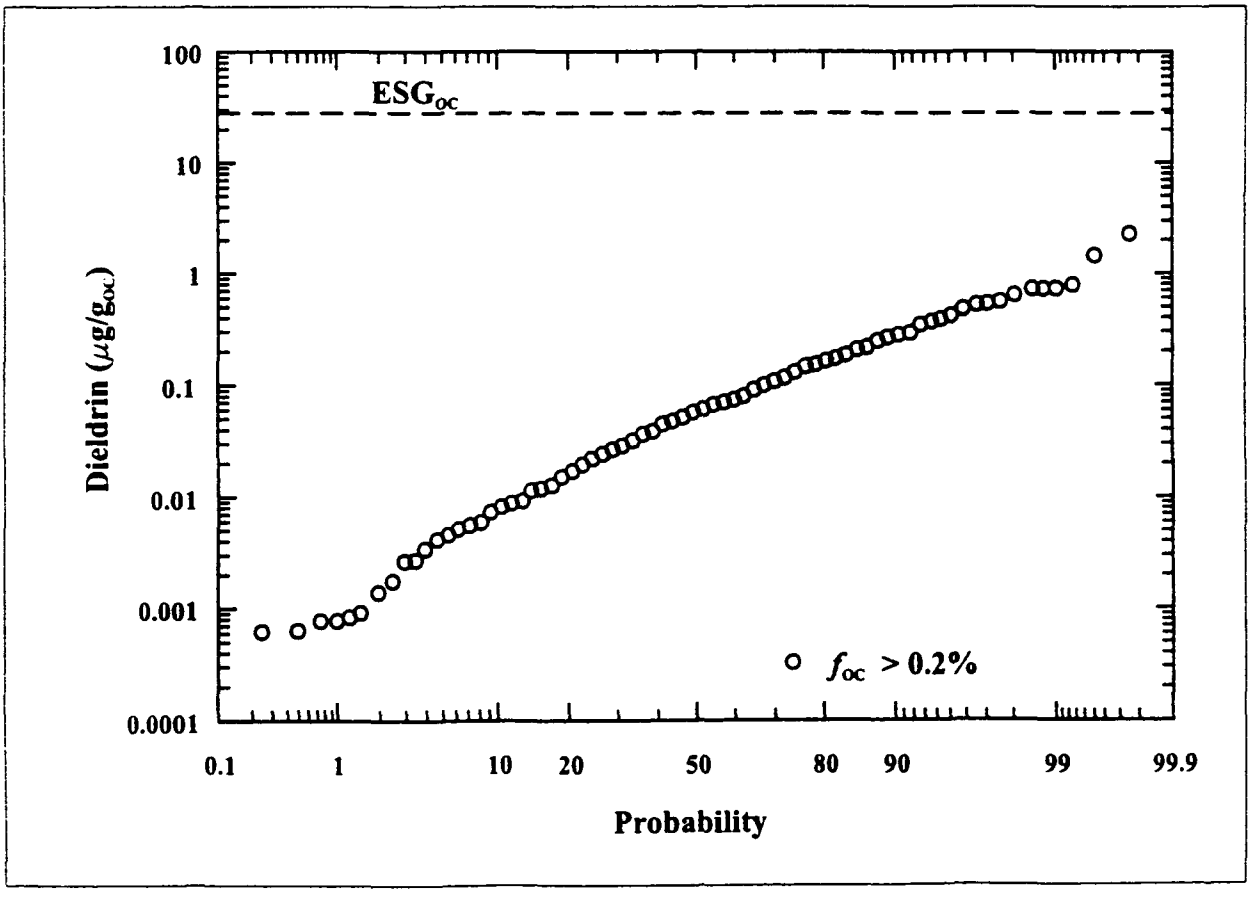


Figure 5-4. Probability distribution of concentrations of dieldrin in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal dashed line is the saltwater ESG value of 28 $\mu g/g_{OC}$.

sediments with 0.20% to 31.9% organic carbon. The median organic carbon–normalized dieldrin concentration (0.080 $\mu g/g_{OC}$) was two orders of magnitude below the ESG of 28 $\mu g/g_{OC}$. None of these samples (n=408) exceeded the guidelines. Hence, these results are consistent with the preceding comparison between the marine ESG 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 (2000a). 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 ESG developed using the EqP methodology. A major portion (93%) of the samples analyzed had f_{OC} >0.2%, for which the ESG 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 ESG. 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 guidelines.

Regional-specific differences in dieldrin concentrations may affect the above conclusions concerning expected guidelines 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

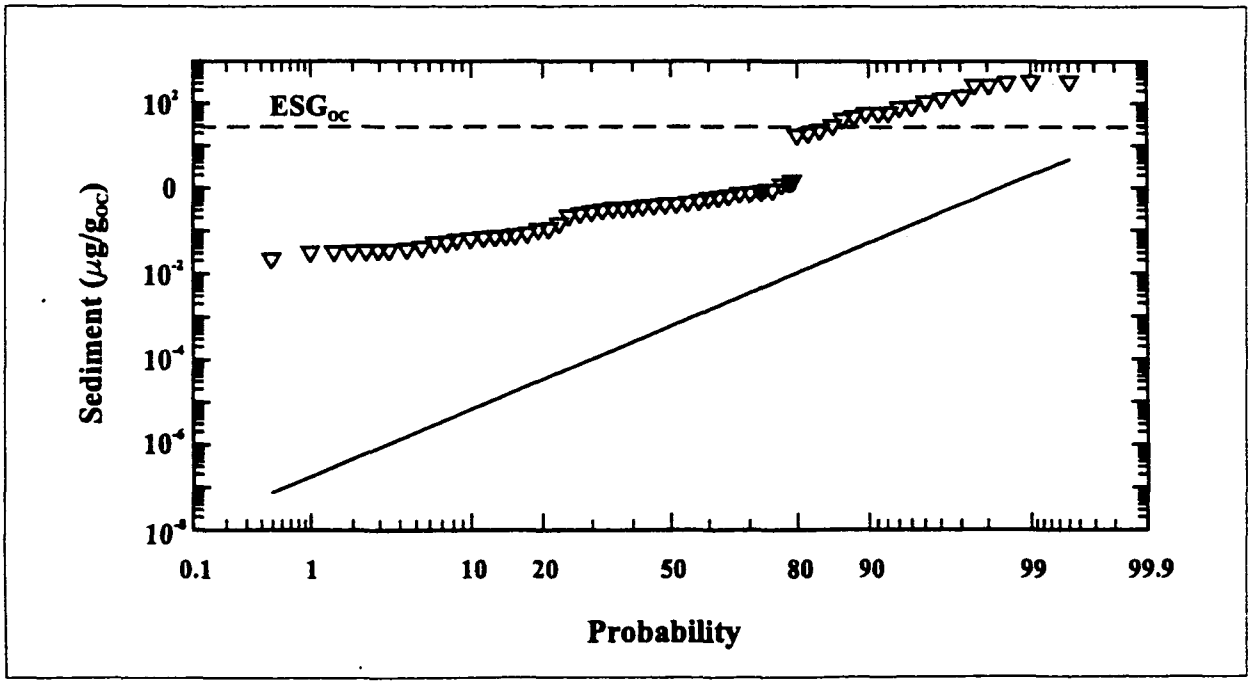


Figure 5-5. **Probability distribution of organic carbon–normalized sediment dieldrin concentrations from the U.S. Army Corps of Engineers (1991) monitoring program of San Francisco Bay. Sediment dieldrin concentrations less than the detection limits are shown as open triangles (▽); measured concentrations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

sampling in different study areas. It is presented as an aid in assessing the range of reported dieldrin sediment concentrations and the extent to which they may exceed the ESG.

5.5 Limitations to the Applicability of ESGs

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 ESG does not mean that other chemicals, for which there are no ESGs available, are not present in concentrations sufficient to cause harmful effects. Furthermore, even if ESGs were available for all of the contaminants in a particular sediment, there might be additive or synergistic effects that the guidelines do not address. In this sense, the ESG represents a "best case" guideline.

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 ESG 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 ESG 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 ESGs exist occur along with ESG chemicals.

Care must be used in applying EqP-derived guidelines in disequilibrium conditions. In some instances, site-specific ESGs may be required to address disequilibrium. The ESGs 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 guidelines 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 guidelines 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 ESG 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 national ESGs would be underprotective and calculation of a site-specific ESG should be considered (see U.S. EPA, 2000b).

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 ESGs 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 ESGs.

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, 2000b).

Section 6

Guidelines Statement

The procedures described in the ESG Technical Basis Document indicate that benthic organisms should be acceptably protected from acute and chronic effects of dieldrin in freshwater sediments containing ≤ 12 μg dieldrin/g_{oc} and saltwater sediments containing ≤ 28 μg dieldrin/g_{oc}, except possibly where a locally important species is very sensitive or sediment organic carbon is <0.2%.

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 are provided as an estimate of the uncertainty associated with the degree to which the observed concentration in sediment ($\mu g/g_{oc}$), which may be toxic, can be predicted using the K_{oc} and the water-only effects concentration. Confidence limits do not incorporate uncertainty associated with water quality criteria. An understanding of the theoretical basis of the equilibrium partitioning methodology, uncertainty, and the partitioning and toxicity of dieldrin are required in the regulatory use of ESGs and their confidence limits.

The guidelines presented in this document are EPA's best recommendation of the concentrations of dieldrin that may be present in sediment while still protecting benthic organisms from the effects of dieldrin. These guidelines 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 guidelines do not protect against additive, synergistic, or antagonistic effects of dieldrin or against the bioaccumulative effects of dieldrin to aquatic life, wildlife, or human health. The Agency and the EPA Science Advisory Board do not recommend the use of ESGs as stand-alone, pass-fail criteria for all applications; rather, exceedances of ESGs could trigger additional studies at sites under investigation. The ESG should be interpreted as a chemical concentration below which adverse effects are not expected. In comparison, at concentrations above the ESG effects are likely, and 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 ESG and associated uncertainty limits.

Section 7

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Appendix A

Summary of Acute Values for Dieldrin for Freshwater and Saltwater Species

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

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Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Scud, *Gammarus fasciatus*	A	E	S	U	4.3	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	X	E	S	U	1.3	—	—	—	Sanders, 1972; Mayer and Ellersieck, 1986
Scud, *Gammarus fasciatus*	X	E	FT	U	5.5	3.133	—	—	Sanders, 1972
Scud, *Gammarus lacustris*	A	E	S	U	3.0	3.0	3.066	3.066	Sanders, 1972; Mayer and Ellersieck, 1986
Glass shrimp, *Palaemonetes kadiakensis*	A	E	S	U	3.2	—	—	—	Sanders, 1972, Mayer and Ellersieck, 1986
Glass shrimp, *Palaemonetes kadiakensis*	X	E	FT	U	0.5	1.265	1.265	1.265	Sanders, 1972; Mayer and Ellersieck, 1986
Crayfish, *Orconectes immunis*	J	E	FT	M	>89	>89	—	—	Thurston et al., 1985
Crayfish, *Orconectes nais*	X	E	S	U	320	—	—	—	Sanders, 1972, Mayer and Ellersieck, 1986
Crayfish, *Orconectes nais*	J	E	S	U	3.2	3.2	3.2	716.88	Sanders, 1972, Mayer and Ellersieck, 1986
Mayfly, *Baetis* sp	J	I	S	U	0.90	0.90	0.90	0.90	Mayer and Ellersieck, 1986
Mayfly, *Hexagenia bilineata*	X	I	S	U	64	—	—	—	Sanders, 1972

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

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

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

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Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0 45	—	—	—	Brungs and Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0 64	0.4899	0 4899	0.4899	Thurston et al., 1985
Black bullhead, *Ictalurus melas*	J	W,E	S	U	1.13	—	—	—	Mayer and Ellersieck, 1986
Black bullhead, *Ictalurus melas*	J	W,E	FT	M	0.45	0.45	—	—	Anderson and DeFoe, 1980
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	0.32	—	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	1.9	—	—	—	Mayer and Ellersieck, 1986
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	0.8	—	—	—	McCorkle et al., 1977
Channel catfish, *Ictalurus punctatus*	J	W,E	FT	M	0.43	—	—	—	Thurston et al., 1985
Channel catfish, *Ictalurus punctatus*	J	W,E	FT	M	0.41	0.4199	0.4347	0.4347	Thurston et al., 1985
Flagfish, *Jordanella floridae*	J	W	FT	M	0.85	0.85	0.85	0.85	Hermanutz, 1978; Hermanutz et al., 1985
Mosquitofish, *Gambusia affinis*	J	W	S	U	1.1	—	—	—	Mayer and Ellersieck, 1986
Mosquitofish, *Gambusia affinis*	X	W	S	U	0.75	—	—	—	Katz and Chadwick, 1961

Common Name, *Scientific Name*	Life-stage[a]	Habitat[b]	Method[c]	Concentration[d]	LC50/EC50[e] (µg/L) Test	HMAV Species[f]	HMAV Genus[g]	Overall GMAV[h]	Reference
Mosquitofish, *Gambusia affinis*	J	W	FT	M	0.69	0 69	0.69	0 69	Thurston et al., 1985
Guppy, *Poecilia reticulata*	X	W	S	U	0 90	—	—	—	Katz and Chadwick, 1961
Guppy, *Poecilia reticulata*	X	W	S	U	1.6	1.200	1 200	1.200	Henderson et al., 1959
Bluegill, *Lepomis macrochirus*	J	W	S	U	0 60	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	8.25	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	5.5	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	2.4	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	1 65	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.86	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.33	—	—	—	Katz and Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0 61	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.41	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.37	—	—	—	Macek et al., 1969; Mayer and Ellersieck, 1986

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

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

Appendix A

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

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

Appendix A

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

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

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

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

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