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Sediment Quality Criteria for the Protection of Benthic Organisms:

ENDRIN

CONTENTS

FOREWORD

Under the Clean Water Act (CWA) the U.S. Environmental Protection Agency (U.S. EPA) and the States develop programs for protecting the chemical, physical, and biological integrity of the nation's waters. Section 304(a)(1) 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, which may be expected from the presence of pollutants in any body of water, including ground water, (2) the concentration and dispersal of pollutants, or their byproducts, through biological, physical and chemical processes, and (3) the effects of pollutants on biological community diversity, productivity, and stability. Section 304(a)(2) directs the Administrator to develop and publish information on, among other things, the factors necessary for the protection and propagation of shellfish, fish, and wildlife for classes and categories of receiving waters.

To meet this objective, U.S. EPA has periodically issued ambient water quality criteria (WQC) guidance beginning with the publication of "Water Quality Criteria 1972" (NAS/NAE, 1973). All criteria guidance through late 1986 was summarized in an U.S. EPA document entitled "Quality Criteria for Water, 1986" (U.S. EPA, 1987). Additional WQC documents that update criteria for selected chemicals and provide new criteria for other pollutants have also been published. In addition to the development of WQC and to continue to comply with the mandate of the CWA, U.S. EPA has conducted efforts to develop and publish sediment quality criteria (SQC) for some of the 65 toxic pollutants or toxic pollutant categories. Section 104 of the CWA authorizes the administrator to conduct and promote research into the causes, effects, extent, prevention, reduction and elimination of pollution, and to publish relevant information. Section 104(n)(1) in particular provides for study of the effects of pollution, including sedimentation in estuaries, on aquatic life, wildlife, and recreation. U.S. EPA's efforts with respect to sediment criteria are also authorized under CWA Section 304(a).

Toxic contaminants in bottom sediments of the nations's lakes, rivers, wetlands, and coastal waters create the potential for continued environmental degradation even where water column contaminant levels meet established WQC. In addition, contaminated sediments can lead to water quality impacts, even when direct discharges to the receiving water have ceased. EPA intends SQC be used to assess the extent of sediment contamination, to aid in implementing measures to limit or prevent additional contamination, and to identify and implement appropriate remediation activities when needed.

The criteria presented in this document are the U.S. EPA's best recommendation of the concentrations of a substance that may be present in sediment while still protecting benthic organisms from the effects of that substance. These criteria are applicable to a variety of freshwater and marine sediments because they are based on the biologically available concentration of the substance in sediments. These criteria do not protect against additive, synergistic or antagonistic effects of contaminants or bioaccumulative effects to aquatic life, wildlife or human health.

The criteria derivation methods outlined in this document are proposed to provide protection of benthic organisms from biological impacts from chemicals present in sediments. Guidelines and guidance are being developed by U.S. EPA to assist in the application of criteria presented in this document, in the development of sediment quality standards, and in other water-related programs of this Agency.

These criteria are being issued in support of U.S. EPA'S regulations and policy initiatives. This document is Agency guidance only. It does not establish or affect legal rights or obligations. It does not establish a binding norm and is not finally determinative of the issues addressed. Agency decisions in any particular case will be made by applying the law and regulations on the basis of the specific facts.

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TABLES

FIGURES

$\mu g/L$) x 1 $Kg_{oc}/1{,}000g_{oc}$). (See Appendix B in this document and Appendix B in the acenaphthene, dieldrin, fluoranthene, and phenanthrene SQC documents for raw data).

Figure 5-1. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to endrin-spiked sediments and sediment concentrations predicted to be chronically safe in fresh water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV), water-only 96-hour lethality tests, Acute Chronic Ratios (ACR) and K_{OC} values. PGMCV = (GMAV ÷ ACR)K_{OC}. Symbols for PGMCVs are Δ for arthropods, ○ for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to endrin-spiked sediments and sediment concentrations predicted to be chronically safe in salt water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV), water-only 96-hour lethality tests, Acute Chronic Ratios (ACR) and K_{OC} values. PGMCV = (GMAV ÷ ACR)K_{OC}. Symbols for PGMCVs are Δ for arthropods, ○ for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

Figure 5-3. Probability distribution of concentrations of endrin in sediments from streams, lakes and estuaries in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared to the endrin SQC values of 0.42 $\mu g/g$ in freshwater sediments having TOC = 10% and 0.042 $\mu g/g$ in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 0.076 $\mu g/g$ when TOC =10% and 0.0076 $\mu g/g$ when TOC=1%. The upper dashed line on each figure represents the SQC value when TOC = 10%, the lower dashed line represents the SQC when TOC = 1%.

DISCLAIMER

This report has been reviewed by the Health and Ecological Criteria Division, Office of Science and Technology, U.S. Environmental Protection Agency, and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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SECTION 1.

INTRODUCTION

1.1 GENERAL INFORMATION

Under the Clean Water Act (CWA) the U.S. Environmental Protection Agency (U.S. EPA) is responsible for protecting the chemical, physical and biological integrity of the nation's waters. In keeping with this responsibility, U.S. 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 the U.S. EPA's best estimate of concentrations protective of human health and the presence and uses of aquatic life. While these water quality criteria play an important role in assuring 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 quality criteria (SQC) makes it difficult to accurately assess the extent of the ecological risks of contaminated sediments and to identify, prioritize and implement appropriate clean up activities and source controls. As a result of the need for a procedure to assist regulatory agencies in making decisions concerning contaminated sediment problems, a U.S. EPA Office of Science and Technology, Health and Ecological Criteria Division (OST/HECD) research team was established 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 SQC derivation in all situations (U.S. EPA, 1989a). The equilibrium partitioning (EqP) approach was selected for non-ionic organic chemicals because it presented the greatest promise for generating defensible national numerical chemical-specific SQC applicable across a broad range of sediment types. The three principal observations that underlie the EqP method of establishing sediment quality criteria are:

1. The concentrations of nonionic organic chemicals in sediments, expressed on an organic carbon basis, and in pore waters correlate to observed biological effects on sediment dwelling organisms across a range of sediments.
2. Partitioning models can relate sediment concentrations for nonionic organic chemicals on an organic carbon basis to freely dissolved concentrations in pore water.
3. The distribution of sensitivities of benthic and water column organisms to chemicals are similar; thus, the currently established WQC final chronic values (FCV) can be used to define the acceptable effects concentration of a chemical freely-dissolved in pore water.

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

The data that support the EqP approach for deriving SQC for nonionic organic chemicals are reviewed by Di Toro et al. (1991) and U.S. EPA, (1993a). Data supporting these observations for endrin are presented in this document.

SQC generated using the EqP method 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; and

5. protective of benthic organisms.

As is the case with WQC, the SQC reflect the use of available scientific data to: 1) assess the likelihood of significant environmental effects to benthic organisms from chemicals in sediments; and 2) to derive regulatory requirements which will protect against these effects.

It should be emphasized that these criteria are intended to protect benthic organisms from the effects of chemicals associated with sediments. SQC are intended to apply to sediments permanently inundated with water, intertidal sediment and sediments inundated periodically for durations sufficient to permit development of benthic assemblages. They do not apply to occasionally inundated soils containing terrestrial organisms. These criteria do not address the question of possible contamination of upper trophic level organisms or the synergistic, additive or antagonistic effects of multiple chemicals. SQC addressing these issues may result in values lower or higher than those presented in this document. The SQC presented in this document represent the U.S. EPA's best recommendation at this time of the concentration of a chemical in sediment that will not adversely affect most benthic organisms. SQC values may be adjusted data or site-specific considerations.

SQC values may also need to be adjusted because of site specific consideration. In spill situations, where chemical equilibrium between water and sediments has not yet been reached, a sediment chemical concentration less than SQC 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 TOC in the sediment does not effect 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 SQC 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 SQC would be overly protective of benthic organisms and should not be used unless modified using the procedures outlined in the "Guidelines for Deriving Site-specific Sediment Quality Criteria for the Protection of Benthic Organisms" (U.S. EPA, 1993b). The SQC may be underprotective where the toxicity of other chemicals are additive with the SQC chemical or species of unusual sensitivity occur at the site.

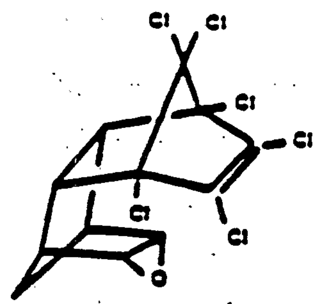
This document presents the theoretical basis and the supporting data relevant to the derivation of the SQC for endrin. An understanding of the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (Stephan et al., 1985), response to public comment (U.S. EPA, 1985) and "Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms By Using Equilibrium Partitioning" (U.S. EPA, 1993a) is necessary in order to understand the following text, tables and calculations. Guidance for the acceptable use of SQC values is contained in "Guide for the Use and Application of Sediment Quality Criteria for Nonionic Organic Chemicals" (U.S. EPA, 1993c).

1.2 GENERAL INFORMATION: ENDRIN

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

Endrin is a cyclic hydrocarbon having a chlorine substituted methanobridge structure (Figure 1-1). Chemically, it is the endo-endo stereoisomer of dieldrin, and has similar physico-chemical properties, except that it is more easily degraded in the environment (Wang, 1988). Endrin is a colorless crystalline solid at room temperature, with a melting point of about 235°C and specific gravity of 1.7 at 20°C. Its vapor pressure is 0.026 mPa (25°C), aqueous solubility approximately 0.024 mg/L at 25°C, and as discussed subsequently, its log octanol-water partition coefficient ($\log_{10}K_{OW}$) is estimated to be 4.90.

Endrin is toxic to non-target aquatic organisms, birds, bees and mammals (Hartley and Kidd, 1987). The acute toxicity of endrin ranges from 0.08 to 352 μg/L for freshwater and 0.037 to 790 μg/L for saltwater organisms (Appendix A). There is little difference between the acute and chronic toxicity of endrin to aquatic species; acute-chronic ratios range from 1.9 to 4.7 for three species (Table 3-3). Endrin bioconcentrates in aquatic animals from 1,450 to 10,000 times the concentration in water (U.S. EPA, 1980). The water quality criterion for endrin (U.S. EPA, 1980) is derived using a Final Residue Value calculated using bioconcentration data and the FDA action level to protect marketability of fish and shellfish; therefore, the WQC is not "effects based". The Final Chronic Value (FCV) in the endrin WQC document (U.S. EPA, 1980) is the recommended concentration protective from direct effects of endrin on aquatic life. This value is modified in this SQC document, and used to derive the SQC.



MOLECULAR FORMULA	$C_{12}H_8C1_60$
MOLECULAR WEIGHT	380.93
DENSITY	1.70 g/cc
MELTING POINT	235°C
PHYSICAL FORM	Colorless crystal
VAPOR PRESSURE	0.026 mPa (25°C)

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

FIGURE 1-1. Chemical structure and physical-chemical properties of endrin.

1.3 OVERVIEW OF DOCUMENT:

Section 1 provides a brief review of the EqP methodology, and a summary of the physical-chemical properties and aquatic toxicity of endrin. Section 2 reviews a variety of methods and data useful in deriving partition coefficients for endrin and includes the K_{OC} recommended for use in the derivation of the endrin SQC. Section 3 reviews aquatic toxicity data contained in the endrin WQC document (U.S. EPA, 1980) and new data that were used to derive the FCV used in this document to derive the SQC concentration. In addition, the comparative sensitivity of benthic and water column species is examined as the justification for the use of the FCV for endrin in the derivation of the SQC. Section 4 reviews data on the toxicity of endrin in sediments, the need for organic carbon normalization of endrin sediment concentrations and the accuracy of the EqP prediction of sediment toxicity using K_{OC} and an effect concentration in water. Data from Sections 2, 3 and 4 are used in Section 5 as the basis for the derivation of the SQC for endrin and its uncertainty. The SQC for endrin is then compared to STORET (U.S. EPA, 1989b) data on endrin's environmental occurrence in sediments. Section 6 concludes with the criteria statement for endrin. The references used in this document are listed in Section 7.

SECTION 2.

PARTITIONING

2.1 DESCRIPTION OF THE EQUILIBRIUM PARTITIONING METHODOLOGY:

Sediment quality criteria (SQC) are the numerical concentrations of individual chemicals which 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 consequence, they can be used in much the same way as water quality criteria (WQC); ie., the concentration of a chemical which is protective of the intended use such as aquatic life protection. For nonionic organic chemicals, SQC are expressed as μg chemical/g organic carbon and apply to sediments having $\geq$ 0.2 % organic carbon by dry weight. A brief overview follows of the concepts which underlie the equilibrium partitioning (EqP) methodology for deriving SQC. The methodology is discussed in detail in the "Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning" (U.S. EPA, 1993a), hereafter referred to as the SQC 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 a SQC 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/liter pore water) and not to 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 pore 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 SQC. It is for this reason that the methodology described below is called the equilibrium partitioning (EqP) method.

It is shown in the SQC Technical Basis Document (U.S. EPA, 1993a) that the final acute values (FAVs) in the WQC documents are appropriate for benthic species for a wide range of chemicals. (The data showing this for endrin are presented in Section 3). Thus, SQC can be established using the final chronic value (FCV) derived using the WQC Guidelines (Stephan et al., 1985) as the acceptable effect concentration in pore or overlying water (see Section 5), and the partition coefficient can be used to relate the pore water concentration to the sediment concentration via the partitioning equation. This acceptable concentration in sediment is the SQC.

The calculation is as follows: Let FCV (μg/L) be the acceptable concentration in water for the chemical of interest; then compute the SQC using the partition coefficient, (K_P) ($L/Kg_{sediment}$), between sediment and water:

$$SQC = K_P \bullet FCV \tag{2-1}$$

This is the fundamental equation used to generate the SQC. Its utility depends upon the existence of a methodology for quantifying the partition coefficient, 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 pore water and sediments (Di Toro et al., 1991). The evidence for endrin is discussed in this section and in section 4. The organic carbon binding of a chemical in sediment is a function of that chemical's organic carbon partition coefficient (K_{OC}) and the weight fraction of organic carbon in the sediment (f_{OC}). The relationship is as follows:

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

It follows that:

$$SQC_{OC} = K_{OC} \bullet FCV \qquad (2\text{-}3)$$

where SQC_{OC} is the sediment quality criterion on a sediment organic carbon basis.

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

2.2 DETERMINATION OF K_{OW} FOR ENDRIN:

Several approaches have been used to determine K_{OW} for the derivation of a SQC, as discussed in the SQC Guidelines. At the U.S. EPA, Environmental Research Laboratory at

Athens, GA (ERL,A) three methods were selected for measurement and two for estimation of K_{OW} values. The measurement methods were shake-centrifugation (SC), generator column (GCol), slow-stir-flask (SSF), and the estimation methods were SPARC (SPARC Performs Automated Reasoning in Chemistry; Karickhoff et al., 1989) and CLOGP (Chou and Jurs, 1979). Data were also extracted from the literature. The SC method is a standard procedure in the Organization for Economic Cooperation and Development (OECD) guidelines for testing chemicals, therefore, it has regulatory precedence.

In the examination of the literature data for endrin, primary references were found listing measured $\log_{10}K_{OW}$ values ranging from 4.40 to 5.19 (Table 2-1). Two primary references were found for estimated values in the literature, 3.54 and 5.6 (Table 2-1). The range of reported values for endrin is significantly greater than the range of values for some other compounds.

TABLE 2-1. ENDRIN MEASURED AND ESTIMATED $LOG_{10}K_{OW}$ VALUES.

METHOD	$LOG_{10}K_{OW}$	REFERENCE
Measured	4.40	Rapaport and Eisenreich, 1984
Measured	4.92	Ellington and Stancil, 1988
Measured	5.01	Eadsforth, 1986
Measured	5.19	De Bruijn et al., 1989
Estimated	3.54	Mabey et al., 1982
Estimated	5.40	SPARC[a]
Estimated	5.60	Neeley et al., 1974

[a]SPARC is from SPARC Performs Automated Reasoning in Chemistry, (Karickhoff et al., 1989).

A K_{OW} value for SPARC is also included in Table 2-1. SPARC is a computer expert system under development at ERL,A, and the University of Georgia, at Athens. For more information on SPARC see U.S. EPA (1993a). The SPARC estimated $\log_{10}K_{OW}$ value for endrin is 5.40.

We had little confidence in the available measured or estimated values for K_{OW}, therefore the SC, GCol, SSF methods were used to provide additional data from which to define K_{OW} for endrin (Table 2-2). The SC method yielded a $\log_{10}K_{OW} = 4.80$ (n=8), the GCol method yielded a $\log_{10}K_{OW} = 4.87$ (n=5), and the SSF method yielded a $\log_{10}K_{OW} = 4.92$ (n=12). Comparison of the results from the SC, GCol, SSF and SPARC K_{OW} determination methods for the five chemicals for which SQC are currently being developed (acenaphthene, dieldrin, endrin, fluoranthene and phenanthrene) indicate that the SSF method provides the best estimate of K_{OW} (U.S. EPA, 1993a). The SSF method had less variability, less experimental bias (Bias is defined as the mean difference between the best-fit estimate of K_{OW} using all four methods and the estimates from each method.) and was generally in the range of the SC, GCol, and SPARC methods (U.S. EPA, 1993a). Therefore, the SSF value of 4.92 is the value for $\log_{10}K_{OW}$ recommended for SQC derivation. This value agrees with the SPARC estimated value and the average of the values measured by the three methods under carefully controlled conditions at ERL,A. This K_{OW} is the logarithm of the mean of 12 K_{OW} measurements made by SSF. The logs of the K_{OW} values measured by SSF ranged from 4.59 to 5.07.

A problem associated with all of the laboratory measurements of endrin is its breakdown on the gas chromatographic column during analysis in each of the phases after equilibrium. Even after the installation of a new column and injection inlet system, breakdown of endrin was observed. To mitigate the effects of this problem, endrin was measured in each phase by adding the areas of the aldehyde and ketone products to the area of the endrin in the chromatogram.

TABLE 2-2. SUMMARY OF $LOG_{10}K_{OW}$[a] VALUES FOR ENDRIN MEASURED BY THE U.S. EPA, ENVIRONMENTAL RESEARCH LABORATORY, ATHENS, GA.

SHAKE-CENTRIFUGATION (SC)	GENERATOR COLUMN (GCol)	SLOW STIR FLASK (SSF)
4.65	4.67	4.86
4.91	5.01	4.59
4.79	4.73	4.97
4.76	4.62	4.95
4.84	5.09	5.02
4.83	5.28[a]	4.82
4.84		5.04
4.83		4.91
		5.07
		4.93
		4.96
		4.78
Mean[b] 4.80[b]	4.87[b]	4.92[b]

[a]Value considered outlier and omitted from mean computation.
[b]Log_{10} of mean of measured values.

2.3 DERIVATION OF K_{OC} FROM ADSORPTION STUDIES:

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

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 a particle interaction effect. This effect suppresses the partition coefficient relative to that observed for undisturbed sediments (Di Toro, 1985; Mackay and Powers, 1987).

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

$$K_p = \frac{f_{OC}\, K_{OC}}{1 + m f_{OC}\, K_{OC} / \upsilon_X} \tag{2-4}$$

where m is the particle concentration in the suspension (kg/L), and $\upsilon_X = 1.4$, an empirical constant. In this expression the K_{OC} is given by:

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

Figure 2-1 compares observed partition coefficient data for the reversible component with calculated values estimated with the particle interaction model (Equation 2-4 and Equation 2-5) for a wide range of compounds (Di Toro, 1985). The endrin datum (Sharom et al., 1980) is highlighted on this plot. The observed $\log_{10} K_p$ of 2.04 reflects significant particle interaction effects. The observed partition coefficient is about nine times lower than the value expected in the absence of particle effects (i.e. $\log_{10} K_p = 2.98$ from $f_{OC} K_{OC} = 958$ L/kg). K_{OC} was computed from Equation 2-5).

In the absence of particle effects, K_{OC} is related to K_{OW} via Equation 2-5. For $\log_{10} K_{OW} = 4.92$ (see section 2.2), this expression results in an estimate of $\log_{10} K_{OC} = 4.84$.

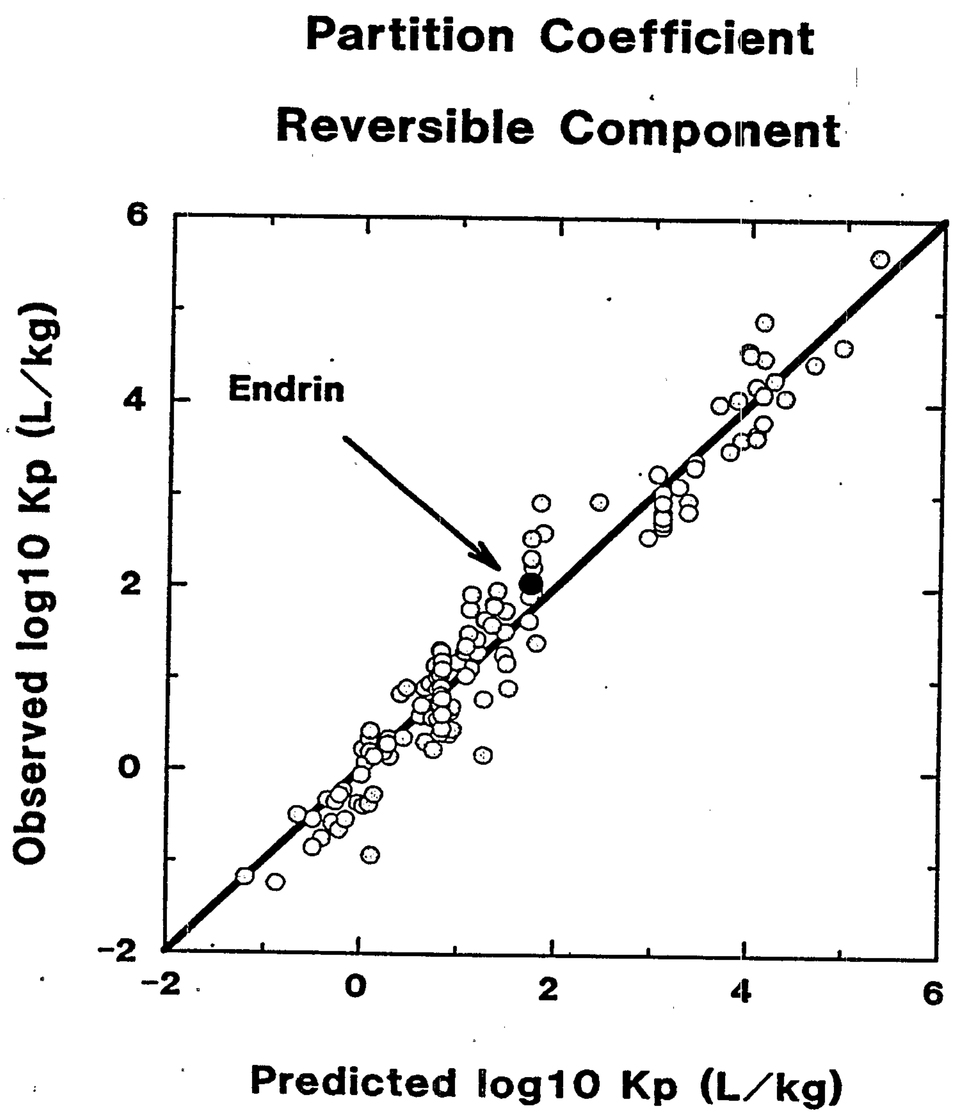


Figure 2-1. Observed versus predicted (equation 2-4) partition coefficients for nonionic organic chemicals (endrin datum is highlighted).

2.3.2 K_{OC} FROM SEDIMENT TOXICITY TESTS:

Measurements of K_{OC} are available from the sediment toxicity tests using endrin (Nebeker et al., 1989; Schuytema et al., 1989; Stehly, 1991). These tests were with different freshwater sediments having a range of organic carbon contents of 0.07 to 11.2 percent (Table 4-1; Appendix B). Endrin concentrations were measured in the sediment and pore waters providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. In the case of the data reported by Schuytema et al. (1989), the concentration of endrin in the overlying water at the end of the 10-day experiment was used. Nebeker et al. (1989) demonstrated using their methodology, that overlying water and pore water endrin concentrations were similar.

The upper panel of Figure 2-2 is a plot of the organic carbon-normalized sorption isotherm for endrin, where the sediment endrin concentration ($\mu g/g_{OC}$) is plotted versus (dissolved) pore water concentration ($\mu g/L$). The data used to make this plot are included in Appendix B. The line of unity slope corresponding to the $\log_{10}K_{OC} = 4.84$ derived from SSF is compared to the data. A probability plot of the observed experimental $\log_{10}K_{OC}$ values is shown in the lower panel of Figure 2-2. The $\log_{10}K_{OC}$ values are approximately normally distributed with a mean of $\log_{10}K_{OC} = 4.67$ and a standard error of the mean of 0.036. This value agrees with the $\log_{10} K_{OC} = 4.84$, which was computed from the SSF determined (Section 2.2) endrin $\log_{10}K_{OW}$ of 4.92 using Equation 2-5.

2.4 SUMMARY OF DERIVATION OF Koc FOR ENDRIN:

The K_{OC} selected to calculate the SQC for endrin is based on the regression of $\log_{10}K_{OC}$

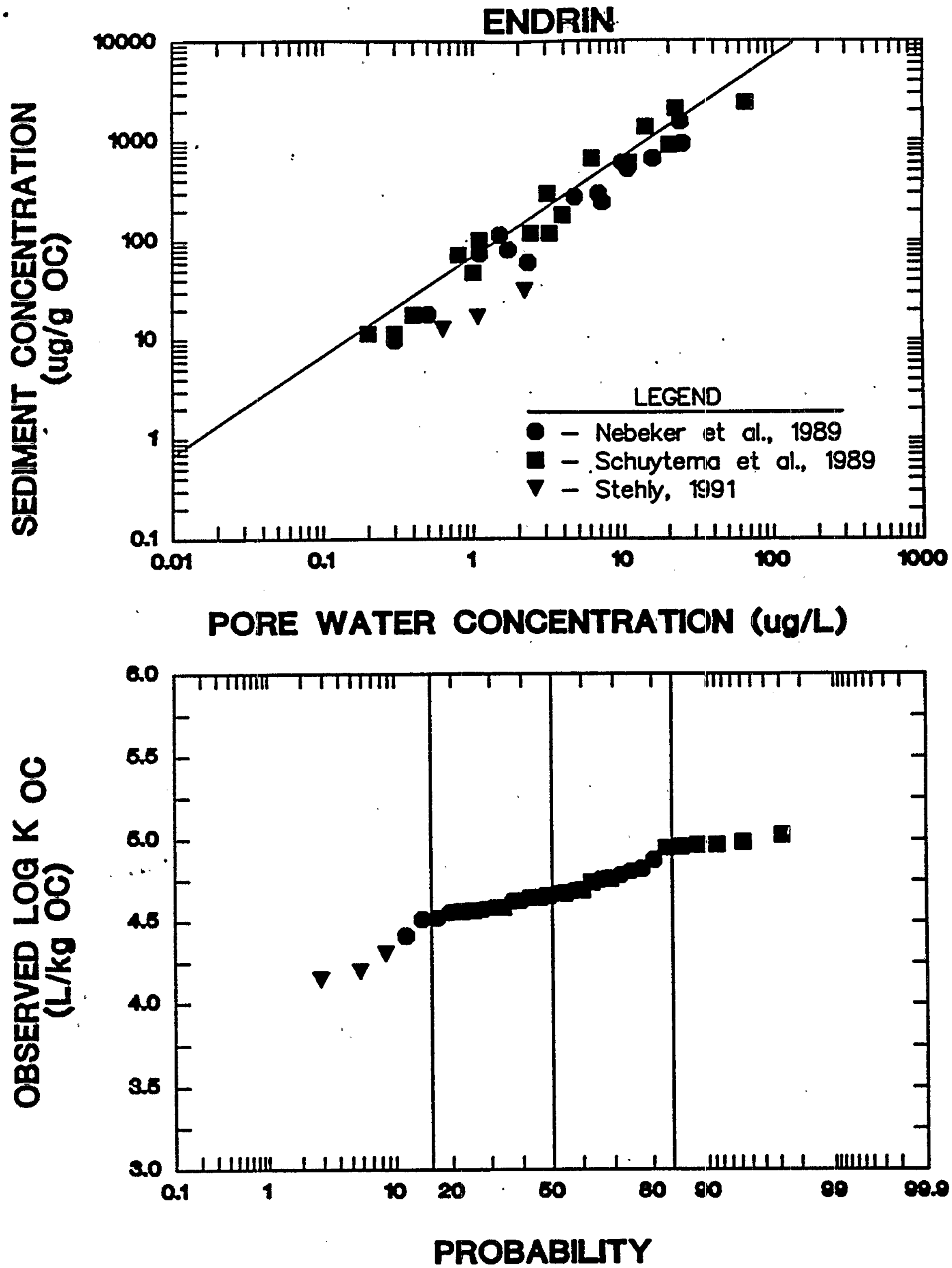


Figure 2-2. Organic carbon-normalized sorption isotherm for endrin (top) and probability plot of K_{OC} (bottom) from sediment toxicity tests conducted by Nebeker et. al, (1989), Schuytema et al. (1989) and Stehly (1991). The line in the top panel represents the relationship predicted with a log K_{oc} of 4.84, that is $C_{s,oc}=K_{oc} \bullet C_d$.

to $\log_{10}K_{OW}$ (Equation 2-5), using the endrin $\log_{10}K_{OW}$ of 4.90 recently measured by ERL,A. This approach rather than the use of the K_{oc} from the toxicity tests was adopted because the regression equation is based on the most robust data set available that spans a broad range of chemicals and particle types, thus encompassing a wide range of K_{OW} and f_{OC}. The regression equation yields a $\log_{10}K_{OC}$ of 4.84. This value is in agreement with the $\log_{10}K_{oc}$ of 4.67 measured in the sediment toxicity tests.

SECTION 3

TOXICITY OF ENDRIN: WATER EXPOSURES

3.1 TOXICITY OF ENDRIN IN WATER: DERIVATION OF ENDRIN WATER QUALITY CRITERIA

The equilibrium partitioning (EqP) method for derivation of sediment quality criteria (SQC) uses the endrin water quality criterion (WQC) Final Chronic Value (FCV) and partition coefficients (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 (epibenthic and infaunal) organisms. For this document, life stages of species classed as benthic are either species that live in the sediment (infauna) or on the sediment surface (epibenthic) and obtain their food from either the sediment or water column (U.S. EPA, 1989c). In this section (1) the FCV from the endrin WQC document (U.S. EPA, 1980) is revised using new aquatic toxicity test data, and (2) the use of this FCV is justified as the effects concentration for SQC derivation.

3.2 ACUTE TOXICITY - WATER EXPOSURES:

One hundred and one standard acute toxicity tests with endrin have been conducted on 45 freshwater species from 35 genera (Figure 3-1; Appendix A). Overall genus mean acute values (GMAVs) range from 0.15 to >165 μg/L. Fishes, amphipods, ostracods, glass shrimp, mayflies, stoneflies, caddisflies, damselflies and dipterans were most sensitive; overall GMAVs for the most sensitive generation of these taxa range from 0.15 to 4.7 μg/L. Thirty-nine tests on the benthic life-stages of 25 species from 21 genera are contained in this database (Figure

3-1; Appendix A). Benthic organisms were among both the most sensitive, and most resistant freshwater species to endrin. Of the epibenthic species, amphipods, grass shrimp, mayflies, stoneflies, caddisflies, damselflies and dipterans are most sensitive; GMAVs range from 0.25 to 5.9 μg/L. Infaunal species tested included endrin sensitive amphipods, stoneflies, mayflies, dipterans and an ostracod (LC50s range from 0.54 to 4.6 μg/L) the endrin-tolerant mayfly, Hexagenia bilineata (LC50=63 μg/L) and the oligochaete, Lumbriculus variegatus, (LC50>165 μg/L). The Final Acute Value (FAV) derived from the overall GMAVs (Stephan et al. 1985) for freshwater organisms is 0.19 μg/L (Table 3-2).

Thirty-seven acute toxicity tests have been conducted on 21 saltwater species from 19 genera (Appendix A). Overall GMAVs range from 0.037 to 790 μg/L. Fishes and a penaeid shrimp were most sensitive; however, only 7 of 21 species tested were invertebrates. Within this database there are results from 26 tests on benthic life-stages of 14 species from 12 genera (Figure 3-2; Appendix A). Benthic organisms are among both the most sensitive and most resistant saltwater genera to endrin. The most sensitive benthic species is the commercially important pink shrimp, Penaeus duorarum, with a flow-through 96-hour LC50 of 0.037 μg/L based on measured concentrations. Other benthic species for which there are data appear less sensitive; GMAVs range from 0.094 to 12 μg/L. The FAV for saltwater species is 0.033 μg/L (Table 3-2).

3.3 CHRONIC TOXICITY - WATER EXPOSURES:

Life-cycle toxicity tests have been conducted with the freshwater flagfish (Jordanella floridae) and fathead minnow (Pimephales promelas), and the saltwater sheepshead minnow

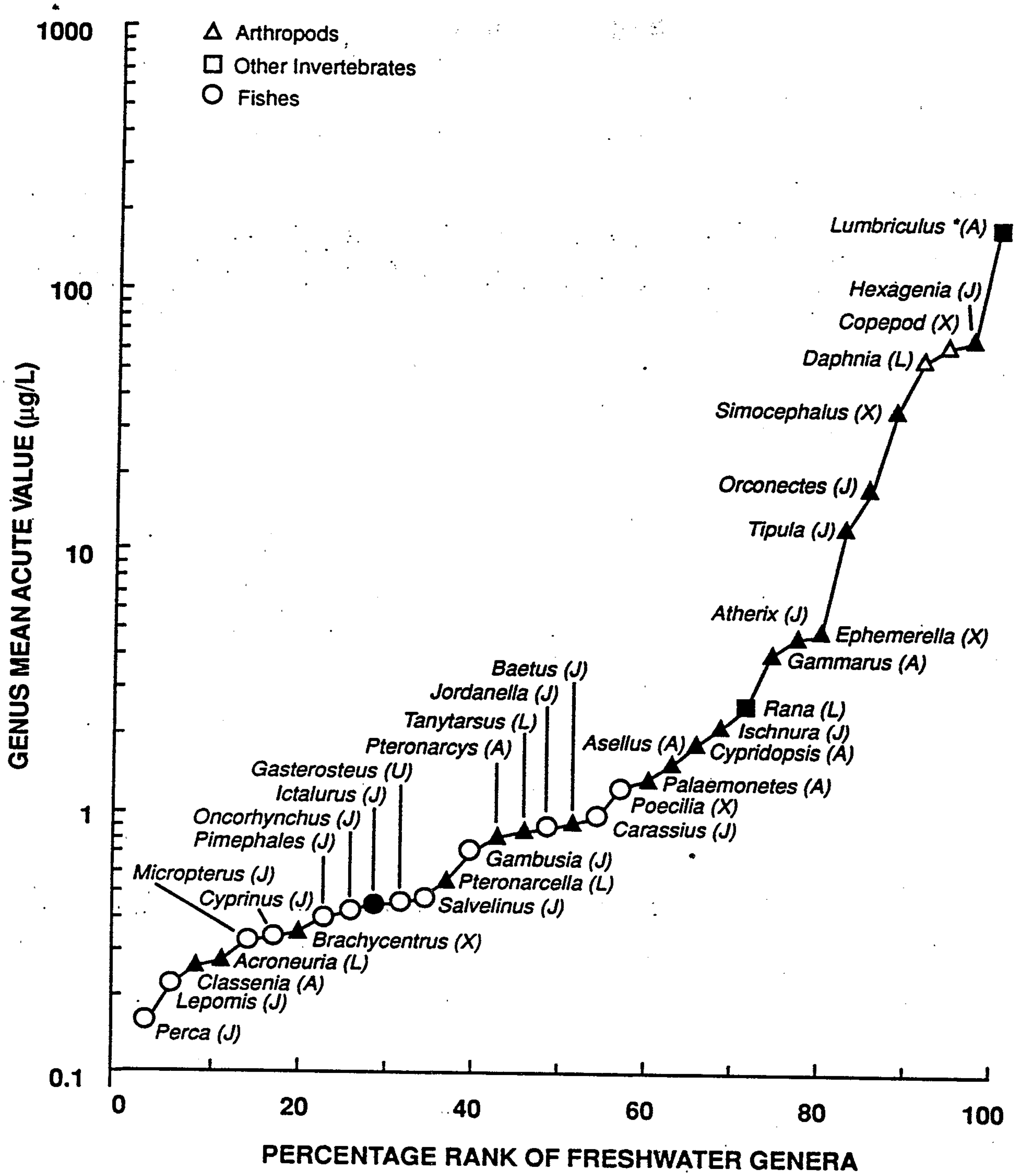


Figure 3-1. Genus mean acute values, of freshwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. A = adult; J = juvenile; U = unspecified life stage, habitat unknown; X = unspecified life stage.

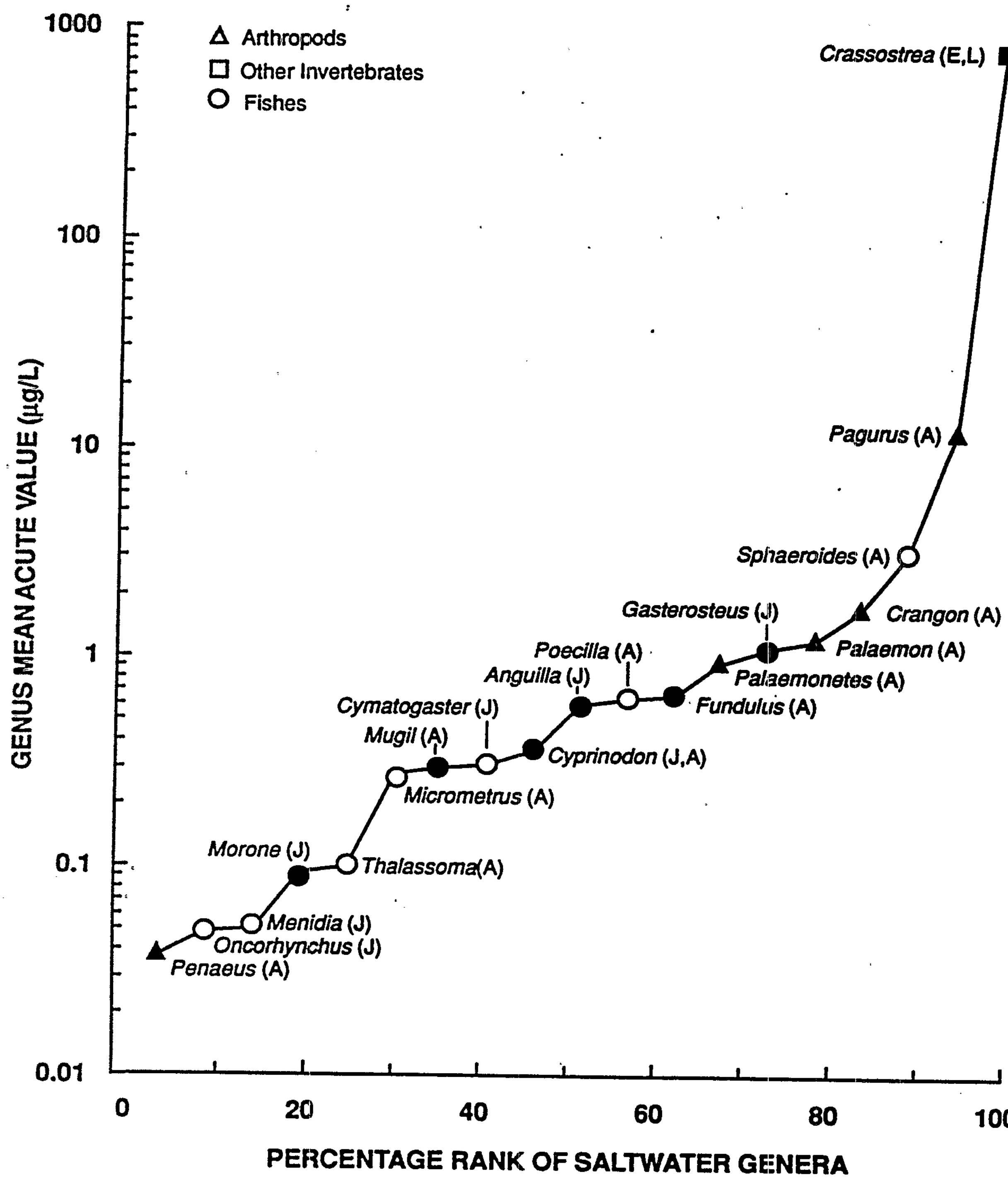


Figure 3-2. Genus mean acute values, of saltwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. E = embryo, J = juvenile.

(Cyprinodon variegatus) and grass shrimp (Palaemonetes pugio) (Table 3-1; 3-2). Each of these species, except for P. promelas, have one or more benthic life stages.

Two life-cycle toxicity tests have been conducted with J. floridae. The concentration-response relationships were almost identical among the tests. Hermanutz (1978) observed an 8% reduction in growth (length) and a 79% reduction in number of eggs spawned per female in 0.3 μg/L endrin relative to response of control fish; progeny were unaffected (Table 3-2). Neither parental or progeny (F_1) generation J. floridae were significantly affected when exposed to endrin concentrations from 0.051 to 0.22 μg/L. In the second life-cycle test, Hermanutz et al. (1985) observed a 51% decrease in reproduction in parental fish exposed to 0.29 μg/L endrin and reductions of 73% in survival, 18% in (growth) length and 92% in numbers of eggs per female in 0.39 μg/L. No significant effects were detected in parental or progeny generation flagfish in 0.21 μg/L.

The effect of endrin on P. promelas in a life-cycle test was only marginally enhanced when exposure was via water and diet vs. water-only exposures (Jarvinen and Tyo, 1978). Parental fish in 0.25 μg/L in water-only exposures exhibited about 60% mortality relative to controls. Mortality of F_1 progeny was 70% in 0.14 μg/L, the lowest concentration tested, and 85% in 0.25 μg/L. Tissue concentrations increased marginally in fish exposed to the water and diet treatment relative to fish in water-only exposures. Effects were observed at all concentrations tested.

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

TABLE 3-1. CHRONIC SENSITIVITY OF FRESHWATER AND SALTWATER ORGANISMS TO ENDRIN. TEST SPECIFIC DATA.

Common Name, Scientific Name	Test[a]	Habitat[b] (Life stage)	Control (CTL) and NOEC[c]	Parental Response LOEC[c]	Parental Response Effect[d]	Progeny Response LOEC[c]	Progeny Response Effect[d]	References
			µg/L	µg/L		µg/L		
				FRESHWATER SPECIES				
Flagfish, *Jordanella floridae*	LC	E(E,L) W(J,A)	CTL, 0.051-0.22	0.30	8%G, 79%R	-	-	Hermanutz, 1978
Flagfish, *Jordanella floridae*	LC	E(E,L) W(J,A)	CTL, 0.21	0.29 0.39	51%R 73%M, 18%G, 92%R	-	-	Hermanutz et al., 1985
Fathead minnow, *Pimephales promelas*	LC	W(E,L, J,A)	CTL < 0.14	0.25	≈60%M	0.14-0.25	70%M 85%M	Jarvinen and Tyo, 1978
				SALTWATER SPECIES				
Grass shrimp, *Palaemonetes pugio*	PLC	W(L) E,W(E, J,A)	CTL, 0.03, 0.05	0.11 0.18 0.38 0.79	38%M 22%M 60%M 100%M	0.11-0.38	26-94%G	Tyler-Schroeder, 1979
Sheepshead minnow, *Cyprinodon variegatus*	LC	E(E) E,W(J,A)	CTL, 0.027-0.12	0.31 0.72	48%M, 15%G, 15%R 100%M	0.31	87%M	Hansen et al., 1977

[a] TEST: LC = lifecycle, PLC = partial lifecycle, ELS = early lifestage
[b] HABITAT: I = infauna, E = epibenthic, W = water column
LIFESTAGE: E = embryo, L = larval, J = Juvenile, A = Adult
[c] NOEC = No observed effect concentration(s); LOEC = lowest observed effect concentration(s).
[d] EFFECT: Percentage decrease relative to controls. M = mortality, G = growth, R = reproduction.

TABLE 3-2. SUMMARY OF FRESH WATER AND SALT WATER ACUTE AND CHRONIC VALUES, ACUTE-CHRONIC RATIOS AND DERIVATION OF FINAL ACUTE VALUES, FINAL ACUTE-CHRONIC RATIOS AND FINAL CHRONIC VALUES FOR ENDRIN.

Common Name, Scientific Name	Acute Value (µg/L)	Chronic Value (µg/L)	Acute-chronic Ratio	Species Mean Acute-chronic Ratio
		FRESHWATER SPECIES		
Flagfish, *Jordanella floridae*	0.85	0.26	3.3	-
Flagfish, *Jordanella floridae*	0.85	0.25	3.4	3.3
Fathead minnow, *Pimephales promelas*	0.41[a]	<0.14	>2.9	>2.9
		SALTWATER SPECIES		
Grass shrimp, *Palaemonetes pugio*	0.35	0.074	4.7	4.7
Sheepshead minnow, *Cyprinodon variegatus*	0.36[b]	0.19	1.9	1.9

[a]Geometric mean of 96 hour LC50 values from five flow-through measured tests (0.26, 0.40, 0.45, 0.49, and 0.50 µg/L) from Brungs and Bailey, 1966 and Solan et al., 1969.

[b]Geometric mean of 96 hour LC50 values from three flow-through measured tests (0.34, 0.37, 0.38 µg/L) on fry or juvenile fish from Hansen et al. 1977 and Schimmel et al., 1975.

Freshwater:

Final Acute Value = 0.19 µg/L
Final Acute-chronic Ratio = 3.1
Final Chronic Value = 0.061 µg/L

Saltwater:

Final Acute Value = 0.033 µg/L
Final Acute-chronic Ratio = 3.1
Final Chronic Value = 0.011 µg/L

delayed, duration of spawning was lengthened and the number of female P. pugio spawning was less in all exposure concentrations from 0.03 to 0.79 μg/L. These effects on reproduction may not be important because embryo production and hatching success were apparently not affected. Larval mortality and time to metamorphosis increased and growth of juvenile progeny decreased in endrin concentrations ≥0.11 μg/L.

C. variegatus exposed to endrin in a life-cycle toxicity test (Hansen et al., 1977) were affected at endrin concentrations similar to those affecting the two freshwater fishes described above. C. variegatus embryos exposed to 0.31 and 0.72 μg/L hatched early; all fry exposed to 0.72 μg/L, and about half those exposed to 0.31 μg/L, died. Females died during spawning, fewer eggs were fertile and survival of exposed progeny decreased in 0.31 μg/L. No significant effects were observed on survival growth or reproduction in fish exposed to 0.027 to 0.12 μg/L.

The difference between the acute and the chronic toxicity of endrin is small (Table 3-2). Acute-chronic ratios (ACR) are 3.3 and 3.4 for J. floridae, 4.7 for P. pugio and 1.9 for C. variegatus. The Final Acute-chronic Ratio (FACR) is 3.1 for both freshwater and saltwater species. Long-term exposures, not classed as "chronic" in the National WQC Guidelines (Stephan et al., 1985), also indicate little difference between the acute and chronic toxicity of endrin. These include tests with caddisfly, Brachycentrus americanus, stonefly, Pteronarcys dorsata (Anderson and DeFoe, 1980) bluntnose minnows, Pimephales notatus (Mount, 1962), fathead minnows, P. promelas (Jarvinen et al., 1988), brown bullheads, Ictalurus melas (Anderson and DeFoe, 1980) largemouth bass, Micropterus salmoides (Fabacher, 1976), spot, Leiostomus xanthurus (Lowe, 1966) and mummichog, Fundulus heteroclitus (Eisler, 1970).

The Final Chronic Values (FCVs) (Table 3-2) are used as the effects concentration for calculating the sediment quality criterion for protection of benthic species. The FCV for freshwater organisms of 0.061 μg/L is the quotient of the FAV of 0.19 μg/L and the FACR of 3.1. Similarly, the FCV for saltwater organisms of 0.011 μg/L is the quotient of the FAV of 0.033 μg/L and the FACR of 3.1.

3.4 APPLICABILITY OF THE WATER QUALITY CRITERION AS THE EFFECTS CONCENTRATION FOR DERIVATION OF THE ENDRIN SEDIMENT QUALITY CRITERION:

The use of the FCV (the chronic effects-based WQC concentration) as the effects concentration for calculation of the EqP-based SQC assumes that benthic (infaunal and epibenthic) species, taken as a group, have sensitivities similar to all species tested to derive the WQC concentration. Data supporting the reasonableness of this assumption over all chemicals for which there are published or draft WQC documents are presented in Di Toro et al. (1991), and the SQC Technical Basis Document (U.S. EPA, 1993a). The conclusion of similarity of sensitivity is supported by comparisons between (1) acute values for the most sensitive benthic 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 vs. all species. Only in this last comparison are endrin-specific comparisons in sensitivity of benthic and all (benthic and water-column) species conducted. The following paragraphs examine the data on the similarity of sensitivity of benthic and all species for endrin.

For endrin, benthic species account for 21 out of 35 genera tested in freshwater, and 12 out of 19 genera tested in saltwater (Figures 3-1, 3-2). An initial test of the difference between the freshwater and saltwater FAVs for all species (water column and benthic) exposed to endrin was performed using the Approximate Randomization method (Noreen, 1989). The Approximate Randomization method tests the significance level of a test statistic when compared to a distribution of statistics generated from many random subsamples. The test statistic in this case is 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 Approximate Randomization method, the freshwater LC50 values and the saltwater LC50 values are combined into one data set. The data set is shuffled, then separated back so that randomly generated "freshwater" and "saltwater" FAVs can be computed. The LC50 values are separated back such that the number of LC50 values used to calculate the sample FAVs are the same as the number used to calculate the original FAVs. These two FAVs are subtracted and the difference used as the sample statistic. This is done many times so that the sample statistics make up a distribution that is representative of the population of FAV differences (Figure 3-3). The test statistic is compared to this distribution to determine it's level of significance. The null hypothesis is that the LC50 values that comprise the saltwater and freshwater data bases are not different. If this is true, the difference between the actual freshwater and saltwater FAVs should be common to the majority of randomly generated FAV differences. For endrin, the test-statistic falls at the 99 percentile of the generated FAV differences. Since the probability is greater than 95%, the hypothesis of no significant difference in sensitivity for freshwater and saltwater species is rejected (Table 3-3).

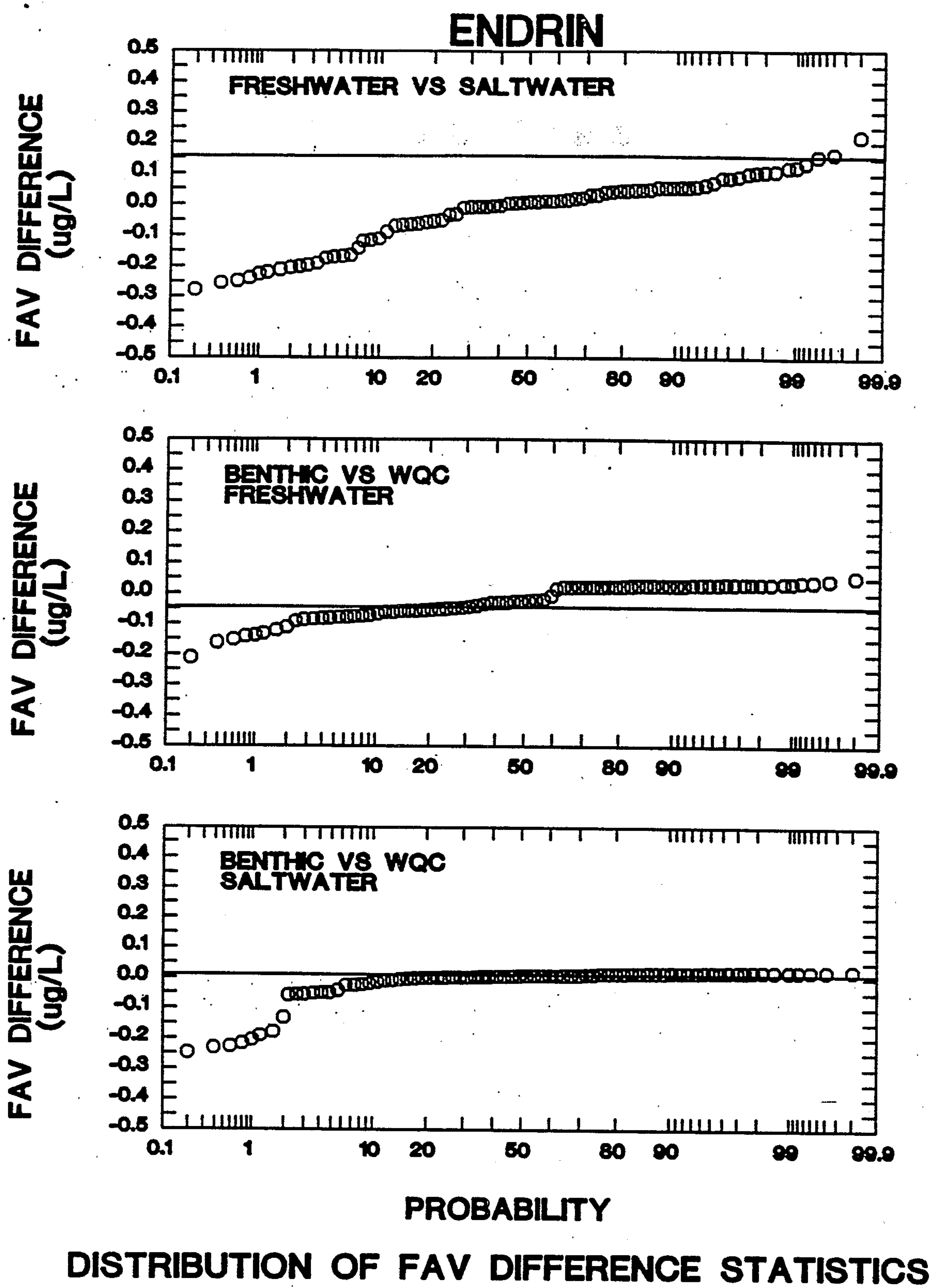


Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater vs. saltwater (upper panel) and benthic vs. WQC (lower panel) data.

TABLE 3-3. RESULTS OF APPROXIMATE RANDOMIZATION TEST FOR THE EQUALITY OF THE FRESHWATER AND SALTWATER FAV DISTRIBUTIONS FOR ENDRIN AND APPROXIMATE RANDOMIZATION TEST FOR THE EQUALITY OF BENTHIC AND COMBINED BENTHIC AND WATER COLUMN (WQC) FAV DISTRIBUTIONS.

Comparison	Habitat or Water Type[a]		AR Statistic[b]	Probability[c]
Fresh vs Salt	Fresh (35)	Salt (19)	.156	99
Fresh: Benthic vs Water Column + Benthic (WQC)	Benthic (21)	WQC (35)	-.045	7
Salt: Benthic vs Water Column + Benthic (WQC)	Benthic (12)	WQC (19)	.010	66

[a]Values in parentheses are the number of LC50 values used in the comparison.
[b]AR statistic = FAV difference between original compared groups.
[c]Pr(AR statistic theoretical ≤ AR statistic observed) given that the samples came from the same population.

Since freshwater and saltwater species do not show similar sensitivity, separate tests were conducted for freshwater and saltwater benthic species. For the species from each water type, a test of difference in sensitivity for benthic and all (benthic and water column species combined, hereafter referred to as "WQC") organisms using the Approximate Randomization method was performed. The test statistic in this case is the difference between the WQC FAV, computed from the WQC LC50 values, and the benthic FAV, computed from the benthic organism LC50 values. This is slightly different than the previous test for saltwater and freshwater species. The difference is that saltwater and freshwater species represent two separate groups. In this test the benthic organisms are a subset of the WQC organisms set. In the Approximate Randomization method for this test, the number of data points coinciding with the number of benthic organisms are selected from the WQC data set. A "benthic" FAV is computed. The original WQC FAV and the "benthic" FAV are then used to compute the difference statistic. This is done many times and the distribution that results is representative of the population of FAV difference statistics. The test statistic is compared to this distribution to determine its level of significance. The probability distribution of the computed FAV differences are shown in the bottom two panels of Figure 3-3. The test statistic for this analysis falls at the 7 percentile for freshwater organisms and the 66 percentile for saltwater organisms. Therefore the hypothesis of no difference in sensitivity is accepted (Table 3-3). This analysis suggests that the FCV for endrin based on data from all tested species is an appropriate effects concentration for benthic organisms.

SECTION 4

TOXICITY OF ENDRIN (ACTUAL AND PREDICTED): SEDIMENT EXPOSURES

4.1 TOXICITY OF ENDRIN IN SEDIMENTS:

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

The only saltwater experiments that tested endrin-spiked sediments were conducted by McLeese et al. (1982) and McLeese and Metcalfe (1980). These began with clean sediments that were added to endrin-coated beakers just prior to the addition of test organisms. This is in marked contrast to tests with freshwater sediments that were spiked with endrin days or weeks prior to test initiation. As a result, the endrin concentrations in the sediment and overlying

TABLE 4-1: SUMMARY OF TESTS WITH ENDRIN SPIKED SEDIMENT.

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method[a]/ Duration (Days)	Response	Sediment Endrin LC50, µg/g Dry wt.	Sediment Endrin LC50, µg/g Org. Car.	Pore Water LC50, µg/L	References
			FRESHWATER SPECIES					
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	1,400	80,000	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	1,050	60,000	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	2,500	143,000	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	2,750[c]	157,000[c]	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	5,400	309,000	-	Keilty et al., 1988a
Tubificid Worm, <u>Limnodrilus hoffmeisteri</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	2,050	117,000	-	Keilty et al., 1988a
Tubificid Worm, <u>Limnodrilus hoffmeisteri</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	3,400	194,000	-	Keilty et al., 1988a
Tubificid Worm, <u>Limnodrilus hoffmeisteri</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	LC50	5,600c	320,000[c]	-	Keilty et al., 1988a
Tubificid Worm, <u>Limnodrilus hoffmeisteri</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	EC50: Sediment avoidance	59.0	3,371	-	Keilty et al., 1988a

TABLE 4-1. Endrin (continued)

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method[a]/ Duration (Days)	Response	Sediment Endrin LC50, µg/g Dry wt.	Sediment Endrin LC50, µg/g Org. Car.	Pore Water LC50, µg/L	References
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	EC50: Sediment avoidance	19.0	1,086	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/4	EC50 Sediment avoidance	15.3	874	-	Keilty et al., 1988a
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/54.2	26% mortality	42.0	2,400	-	Keilty et al., 1988b
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/54.2	18% mortality	11.5	657	-	Keilty et al., 1988b
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/54.2	Weight loss	≥2.33	133	-	Keilty et al., 1988b
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,M/54.2	Decreased sediment reworking?	≥0.54	30.8	-	Keilty et al., 1988b
Lumbriculid worm, <u>Stylodrilus heringianus</u>	Lake Michigan; 0.25mm sieved.	1.75[b]	S,N 69	Weight loss	50.0	2,860	-	Keilty and Stehly, 1989
Amphipod <u>Diporeia</u> sp.	Lake Michigan; depth 29m.	0.07	S,M/4	LC50	0.012	17.0	1.07	Stehly, 1992
Amphipod, <u>Diporeia</u> sp.	Lake Michigan; depth 45m.	0.55	S,M/4	LC50	0.172	31.3	2.2	Stehly, 1992
Amphipod, <u>Diporeia</u> sp.	Lake Michigan; depth 100m.	1.75	S,M/4	LC50	0.224	12.8	0.63	Stehly, 1992
Amphipod, <u>Hyalella azteca</u>	Soap Creek Pond No. 7, OR.	3.0	S,M/10	LC50	4.4	147	2.1	Nebeker et al., 1989

TABLE 4-1. Endrin (continued)

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method[a]/ Duration (Days)	Response	Sediment Endrin LC50, µg/g Dry wt.	Sediment Endrin LC50, µg/g Org. Car.	Pore Water LC50, µg/L	References
Amphipod, *Hyalella azteca*	1:1 mixture of Soap Creek and Mercer Lake, OR.	6.1	S,M/10	LC50	4.8	78.7	1.9	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR.	11.2	S,M/10	LC50	6.0	53.6	1.8	Nebeker et al., 1989
Amphipod, *Hyalella azteca*	Soap Creek Pond No. 7, OR; refrigerated.	3	S,M/10	LC50	5.1	170	-	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Soap Creek Pond No. 7, OR; frozen.	3	S,M/10	LC50	7.7	257	-	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; refrigerated.	11	S,M/10	LC50	19.6	178	-	Schuytema et al; 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; frozen.	11	S,M/10	LC50	21.7	197	-	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; refrigerated.	11	S,M/10	LC50	10.3	93.6	-	Schuytema et al., 1989
Amphipod, *Hyalella azteca*	Mercer Lake, OR; frozen.	11	S,M/10	LC50	9.8	89.1	-	Schuytema et al., 1989
			SALTWATER SPECIES					
Polychaete Worm, *Nereis virens*	17% sand, 83% silt and clay[d].	2	R,M/12	2 of 5 worms died	28	1,400	-	McLeese et al., 1982
Sand Shrimp, *Crangon septemspinosa*	Sand, wet-sieved between 1-2mm sieves[d]	0.28	R,M/4	LC50	0.047	16.8	-	McLeese and Metcalfe 1980

[a]S = Static; R = Renewal; M = Measured; N = nominal.
[b]Value from Landrum (1991).

cFree dieldrin measured by Landrum method.

d*L. hoffmeisteri* and *S. heringianus* tested together.

[e]Clean sediment placed in endrin-coated beakers at beginning of exposure.

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 are uncertain.

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

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

Keilty et al. (1988a) examined the effects of endrin-spiked sediment on sediment avoidance and mortality of two species of oligochaete worms in replicate 4 day exposures (Table 4-1). Four day LC50 values for five tests with Stylodrilus heringianus averaged 2,220 μg endrin/g dry wt. sediment; range 1,050 to 5,400 μg/g. Four day LC50 values for three tests with Limnodrilus hoffmeisteri averaged 3,390 μg/g dry wt. sediment; range 2,050 to 5,600

$\mu g/g$. Four day LC50 values from these tests averaged 194,000 $\mu g/g_{oc}$ for L. hoffmeisteri and 127,000 $\mu g/g_{oc}$ for S. heringianus. Data using this test method suggest within laboratory variabilities of factors of 3 to 5 in LC50 values for the same sediment. Sediment avoidance was seen at much lower concentrations. Over all tests burrowing was markedly reduced at $\geq$ 11.5 $\mu g/g$ and possibly at $\geq$ 0.54 $\mu g/g$. Concentrations where 50% of the worms failed to burrow into sediments (EC50) were 59.0 $\mu g/g$ (3,371 $\mu g/g_{OC}$) for L. hoffmeisteri and 15.3 and 19 $\mu g/g$ dry wt. (874 and 1,086 $\mu g/g_{OC}$) sediment for two tests using S. heringianus. Keilty et al. (1988b) observed 18% mortality of S. heringianus in 11.5 $\mu g/g$ after a 54 day exposures and 26% mortality in 42.0 $\mu g/g$. Sediment reworking rate was reported to be significantly reduced or increased in sediments containing $\geq$ 0.0031 $\mu g/g$. Dry weights of worms in $\geq$2.33 $\mu g/g$ were reduced after 54 days. Keilty and Stehly (1989) observed no effect of a single, nominal concentration of 50 $\mu g/g$ dry wt. sediment on protein utilization by S. heringianus over the 69 day exposure period. However dry weights of worms were significantly reduced.

Nebeker et al. (1989) and Schuytema et al. (1989) in a series of experiments important to the development of sediment quality criteria, exposed the amphipod Hyalella azteca to two of the same endrin-spiked sediments; one with a TOC of 11% and the other 3%. Nebeker et al. (1989) mixed these two sediments to obtain a third sediment with a TOC of 6.1%. Sediments were shaken for 7 days in endrin-coated flasks, and subsequently for 62 days in clean flasks. The 10 day LC50's for amphipods in the three sediments tested by Nebeker et al. (1989) did not differ when endrin concentration was on a wet or dry weight basis. The LC50 values decreased with increase in organic carbon when the concentration was on an organic carbon basis (Table 4-1). The authors conclude that endrin data do not support equilibrium partitioning theory. LC50's normalized to dry weight (4.4 to 6.0 $\mu g/g$) or wet weight (0.9 to 1.0 $\mu g/g$) differed by less than a factor of 1.5 over a 3.7 fold range of TOC. In contrast, the organic carbon normalized LC50s ranged from 53.6 to 147 $\mu g/g_{oc}$, a factor of 2.7 (Table 4-1).

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

The need for organic carbon normalization of the concentrations of nonionic organic chemicals in sediments is presented in the SQC Technical Basis Document (U.S. EPA, 1993a). The need for organic carbon normalization for endrin is supported by the results of spiked-sediment toxicity tests described above. When examined individually, experiments in which H. azteca were exposed to the same sediments by both Nebeker et al. (1989) and Schuytema et al. (1989) provide contradictory data concerning the need for organic carbon normalization (Tables 4-1). Nebeker et al. (1989) observed no change in toxicity and Schuytema et al. (1989) a decrease in toxicity on a dry weight basis. However, mean LC50 values calculated for individual experiments from both studies were similar when concentrations were normalized by organic carbon content. The LC50 was 109 $\mu g/g_{OC}$ (5 tests) for sediments from Mercer Lake having a TOC of 11% and 186 $\mu g/g_{oc}$ (3 tests) for sediments from Soap Creek Pond having 3% organic carbon. The lack of consistent evidence supporting organic carbon normalization in the individual tests (Nebeker et al., 1989), is in contrast with evidence supporting normalization overall tests with the same sediments spiked with endrin, is most likely because organic carbon concentrations differed by less than a factor of four and variability inherent in these tests limited the capacity for discrimination. Sediments tested by Stehly (1992) further provide strong evidence which also supports the need for normalization for endrin (Table 4-1). The organic carbon concentrations for these sediments ranged from 0.07 to 1.75% (a factor of 25). On a dry weight basis, 4-day LC50 values for Diporeia sp. ranged from 0.012 to 0.224 $\mu g/g$ (a factor of 18.7). The organic carbon normalized LC50s were within a factor of 2.4; range 12.8 to 31.3 $\mu g/g_{OC}$.

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

4.2 CORRELATION BETWEEN ORGANISM RESPONSE AND PORE WATER CONCENTRATION:

One corollary of the EqP theory is that pore water LC50's for a given organism should be constant across sediments of varying organic carbon content (U.S. EPA, 1993a). Appropriate pore water LC50 values are available from two studies using endrin (Table 4-1). Nebeker et al. (1989) found 10 day LC50 values for endrin based on pore water concentrations ranged 1.8 to 2.1 μg/L for H. azteca exposed to three sediments. Overlying water LC50 values from these static tests and those conducted using the same sediments by Schuytema et al. (1989) were similar; 1.1 to 3.9 μg/L. Stehly (1992) found that 10 day pore water LC50 values for Diporeia sp. ranged from 0.63 to 2.2 μg/L (a factor of 3.5); this is considerably less than the range in dry wt. LC50's, 0.012 to 0.224 μg/g (a factor of 18.7), for three sediments from Lake Michigan having 0.07 to 1.75% organic carbon.

A more detailed evaluation of the degree to which the response of benthic organisms can be predicted from toxic units of substances in pore water can be made utilizing results from toxicity tests with sediments spiked with other substances, including acenaphthene and phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), dieldrin (Hoke 1992), fluoranthene (Swartz et al., 1990, DeWitt et al., 1992), or kepone (Adams et al., 1985) (Figure 4-1; Appendix B). The data included in this analysis come from tests conducted at EPA laboratories or from tests which utilized designs at least as rigorous as those conducted at the EPA laboratories. Tests with acenaphthene and phenanthrene used two saltwater amphipods (Leptocheirus plumulosus and Eohaustorius estuarius). Tests with fluoranthene used a saltwater amphipod (Rhepoxynius abronius) and marine sediments. Freshwater sediments spiked with endrin were tested using the amphipod (H. azteca); the midge Chironomus tentans was tested using kepone. Figure 4-1 presents the percentage mortalities of the benthic species tested in individual treatments for each chemical versus "pore water toxic units" (PWTUs) for all sediments tested. PWTUs are the concentration of the chemical in pore water (μg/L) divided by the water only LC50 (μ/L). Theoretically, 50% mortality should occur at one interstitial

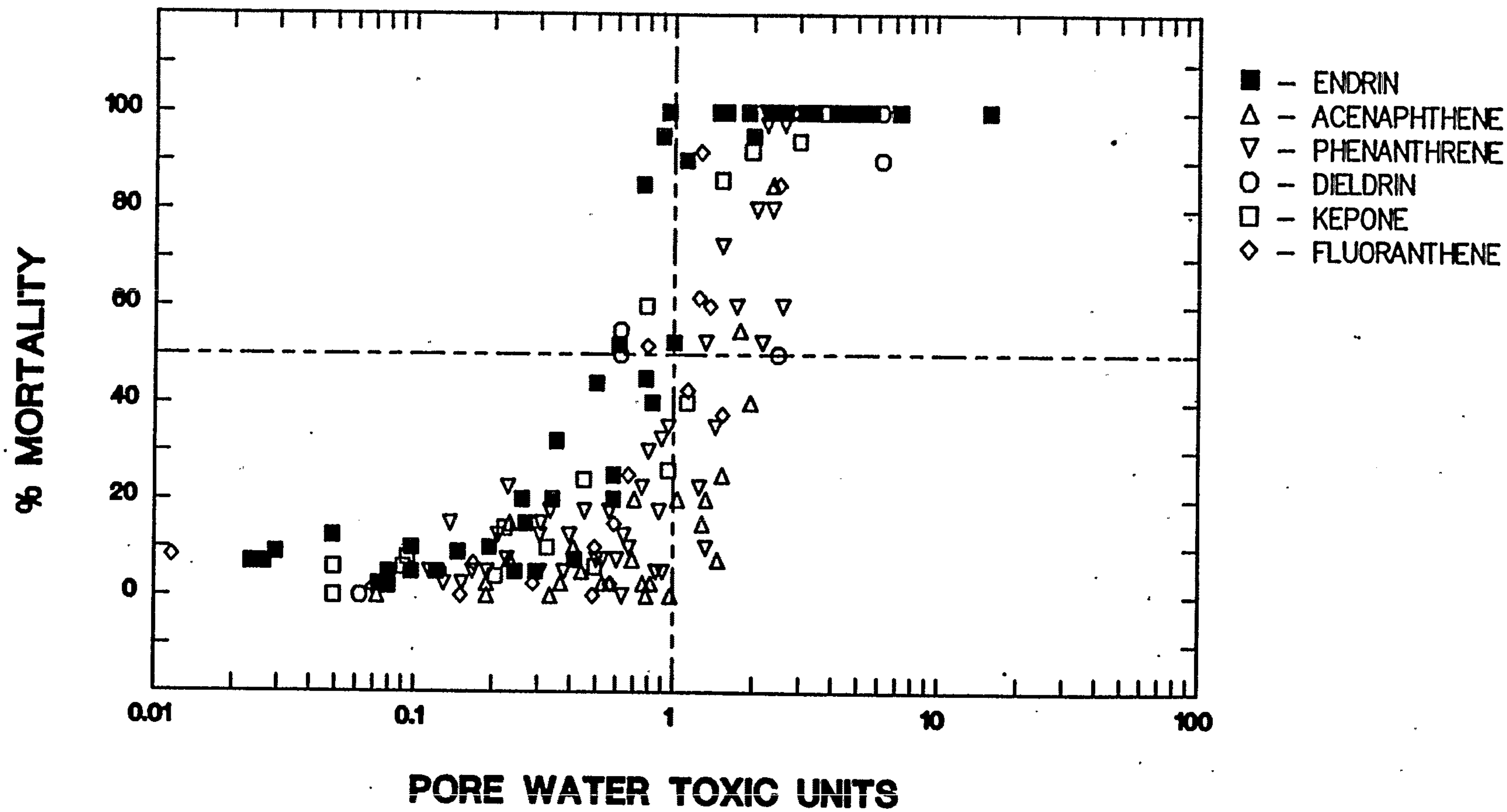


Figure 4-1. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989) or fluoranthene (Swartz et al., 1990), and midge in sediments spiked with dieldrin (Hoke, 1992) or kepone (Adams et al., 1985) relative to pore water toxic units. Pore water toxic units are ratios of concentrations of chemicals measured in individual treatments divided by the water-only LC50 value from water-only tests. (See Appendix B in this SQC document, Appendix B in the acenaphthene, dieldrin, fluoranthene and phenanthrene SQC documents, and original references for raw data.)

water toxic unit. At concentrations below one PWTU there should be less than 50% mortality, and at concentrations above one PWTU there should be greater than 50% mortality. Figure 4-1 shows that at concentrations below one PWTU mortality was generally low, and increased sharply at approximately one PWTU. Therefore, this comparison supports the concept that interstitial water concentrations can be used to predict the response of an organism to a chemical that is not sediment-specific. This concept was not used to derive sediment quality criteria because of the complexation of non-ionic organic chemicals with pore water DOC (Section 2) and the difficulties of adequately sampling pore waters.

4.3 TESTS OF THE EQUILIBRIUM PARTITIONING PREDICTION OF SEDIMENT TOXICITY:

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

Predicted ten-day LC50 values from endrin-spiked sediment tests with H. azteca (Nebeker et al.,1989; Schuytema et al., 1989) were calculated (Table 4-2) using the $Log_{10}K_{OC}$ value of 4.84 from Section 2 of this document and the sediment LC50's (Nebeker et al. 1989) from tests conducted jointly by these authors. Overall, ratios of actual to predicted LC50s for endrin averaged 0.44 (range 0.18 to 0.90) in nine tests with three sediments.

A more detailed evaluation of the accuracy and precision of the EqP prediction of the response of benthic organisms can be made using the results of toxicity tests with amphipods exposed to sediments spiked with acenaphthene, phenanthrene, dieldrin, endrin, or fluoranthene. The data included in this analysis come from tests conducted at EPA laboratories or from tests which utilized designs at least as rigorous as those conducted at the EPA laboratories. Data from the kepone experiments are not included because a measured K_{OW} for kepone obtained using the slow stir flask method is not available. 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 and Ankley (1991) exposed the amphipod Hyalella azteca to three dieldrin-spiked freshwater sediments having 1.7, 3.0 and 8.5% organic carbon and Hoke (1992) 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 percentage mortalities of amphipods in individual treatments of each chemical versus "predicted sediment toxic units" (PSTU) for each sediment treatment. PSTUs are the

TABLE 4-2: WATER-ONLY AND SEDIMENT LC50S USED TO TEST THE APPLICABILITY OF THE EQUILIBRIUM PARTITIONING THEORY FOR ENDRIN.

Common/Sci. Name	Method Duration (days)	Water Only LC50	Overlying Water LC50	Pore Water LC50	TOC (%)	Sediment Endrin LC50		Predicted[a] LC50 µg/g OC	Ratio: Actual LC50+ Predicted LC50	Reference
						µg/g Dry Wt.	µg/g OC			
		µg/L	µg/L	µg/L						
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.2[b]	1.3[b]	2.1[b]	3.0	4.4	147	290	0.51	Nebeker et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	3.8[b]	1.1[b]	1.9[b]	6.1	4.8	78.7	263	0.30	Nebeker et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.3[b]	1.2[b]	1.8[b]	11.2	6.0	53.6	297	0.18	Nebeker et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	1.8[b]	-	3	5.1	170	284	0.60	Schuytema et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	3.6[b]	-	3	7.7	257	284	0.90	Schuytema et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	3.6[b]	-	11	19.6	178	284	0.63	Schuytema et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	3.9[b]	-	11	21.7	197	284	0.69	Schuytema et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	1.4[b]	-	11	10.3	93.6	284	0.33	Schuytema et al., 1989
Amphipod, <u>Hyalella</u> <u>azteca</u>	S,M/10	4.1[c]	1.8[b]	-	11	9.8	89.1	284	0.31	Schuytema et al., 1989
Geometric Mean		4.1[c]	1.9[b]	1.9[b]	-	-	125.8	284[a]	0.44	

[a] Predicted LC50 (µg/g OC) = Water-only LC50 (µg/L) x K_{oc} (L/Kg_{oc}) x $1Kg_{oc}/1000g_{oc}$; where $K_{oc} = 10^{4.84}$

[b] Soluble endrin; samples centrifuged prior to analysis.

[c] Mean 10 day-water only LC50 from 3 tests in Nebeker et al. 1989.

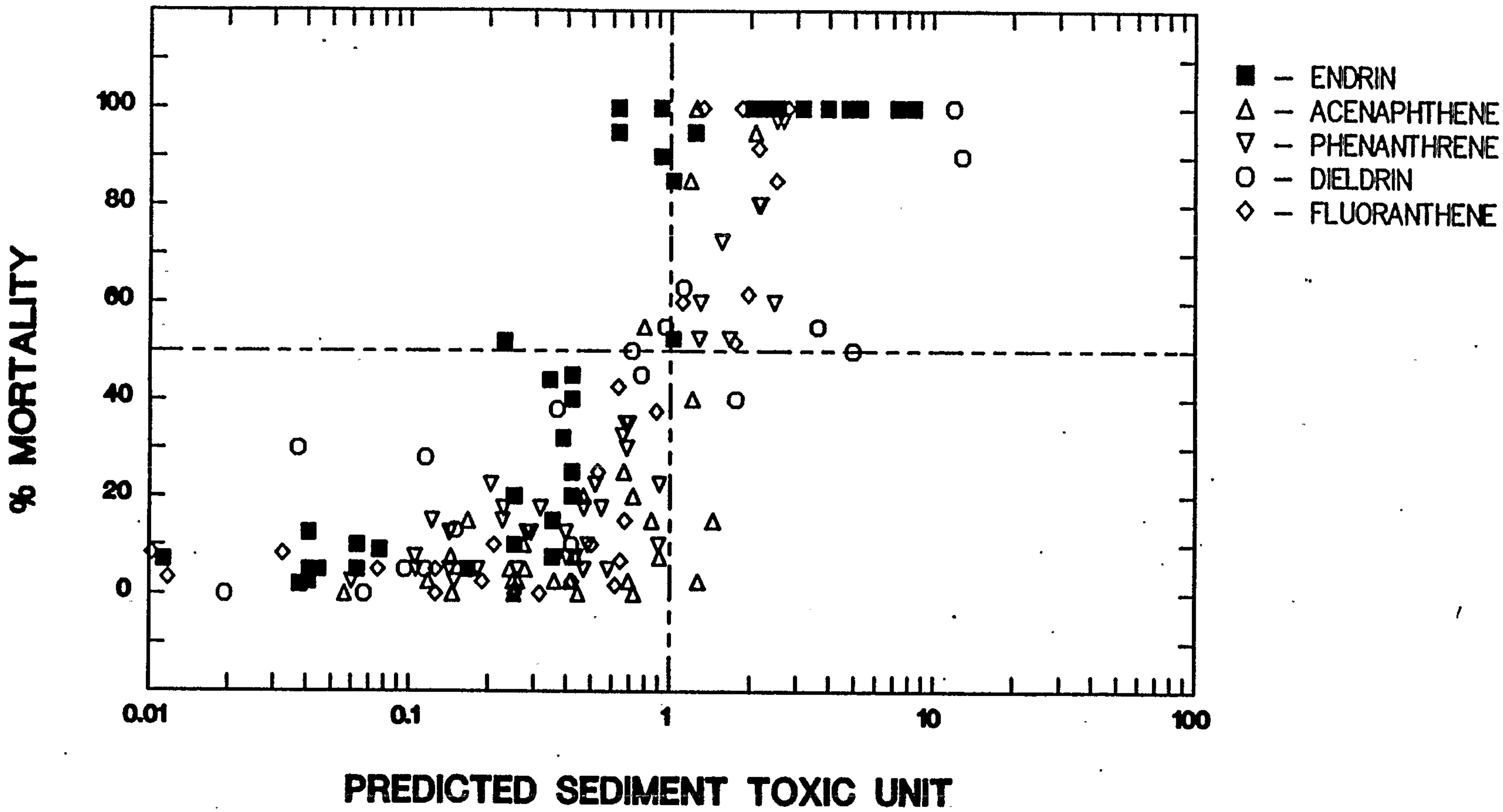


Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke and Ankley, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989) or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992) and midge in dieldrin spiked sediments (Hoke, 1992) relative to "predicted sediment toxic units." Predicted sediment toxic units are the ratios of measured treatment concentrations for each chemical in sediments ($\mu g/g_{OC}$) divided by the predicted LC50 ($\mu g/g_{OC}$) in sediments (K_{OC} x Water Only LC50, $\mu g/L$) x 1 $Kg_{oc}/1{,}000g_{oc}$). (See Appendix B in this document and Appendix B in the acenaphthene, dieldrin, fluoranthene, and phenanthrene SQC documents for raw data).

concentration of the chemical in sediments ($\mu g/g_{OC}$) divided by the predicted LC50 ($\mu g/g_{OC}$) in sediments (the product of K_{OC} and the 10-day water only LC50). In this normalization, 50% mortality should occur at one PSTU. At concentrations below one PSTU mortality was generally low, and increased sharply at one PSTU. The means of the LC50s for these tests calculated on a PSTU basis were 1.90 for acenaphthene, 1.16 for dieldrin, 0.44 for endrin, 0.80 for fluoranthene and 1.22 for phenanthrene. The mean value for the five chemicals is 0.99. This illustrates that the EqP method can account for the effects of different sediment properties and properly predict the effects concentration in sediments using effects concentration from water only exposures.

SECTION 5

CRITERIA DERIVATION FOR ENDRIN

5.1 CRITERIA DERIVATION:

The water quality criteria (WQC) Final Chronic Value (FCV), without an averaging period or return frequency (See section 3), is used calculate the sediment quality criteria (SQC) because it is probable that the concentration of contaminants in sediments are relatively stable over time, thus exposure to sedentary benthic species should be chronic and relatively constant. This is in contrast to the situation in the water column, where a rapid change in exposure and exposures of limited durations can occur due to fluctuations in effluent concentrations, dilutions in receiving waters or the free-swimming or planktonic nature of water column organisms. For some particular uses of the SQC it may be appropriate to use the areal extent and vertical stratification of contamination of a sediment at a 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 Final Acute Value (FAV), and the final Acute Chronic Ratio (ACR) for the substance. The FAV is an estimate of the acute LC50 or EC50 concentration of the substance corresponding to a cumulative probability of 0.05 from eight or more families for the genera for which acceptable acute tests have been conducted on the substance. The ACR is the mean ratio of acute to chronic toxicity for three or more species exposed to the substance that meets minimum database requirements. For more information on the calculation of ACRs, FAVs, and FCVs see the National Water Quality Criteria Guidelines (Stephan et al., 1985). The FCV used in this document differs from the FCV in the endrin WQC document (U.S. EPA, 1980) because it incorporates recent data not included in that document, and omits some data which does not meet the data requirements established in the WQC Guidelines (Stephan et al., 1985).

The equilibrium partitioning (EqP) method for calculating SQC is based on the following procedure. If FCV (μg/L) is the chronic concentration from the WQC for the chemical of interest, then the SQC (μg/g sediment), is computed using the partition coefficient, K_P (L/g sediment), between sediment and pore water:

$$SQC = K_P \, FCV \qquad (5\text{-}1)$$

Since organic carbon is the predominant sorption phase for nonionic organic chemicals in naturally occurring sediments, (salinity, grainsize and other sediment parameters have inconsequential roles in sorption, see sections 2.1 and 4.3) the organic carbon partition coefficient, (K_{OC}) can be substituted for K_P. Therefore, on a sediment organic carbon basis, the SQC_{OC} ($\mu g/g_{OC}$), is:

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

Since (K_{OC}) is presumably independent of sediment type for non-ionic organic chemicals, so also is SQC_{OC}. Table 5-1 contains the calculation of the endrin SQC.

The organic carbon normalized SQC is applicable to sediments with an organic carbon fraction of $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic carbon normalization and SQC may not apply.

TABLE 5-1. SEDIMENT QUALITY CRITERIA FOR ENDRIN.

Type of Water Body	Log K_{OW} (L/kg)	Log K_{OC} (L/kg)	FCV (μg/L)	SQC_{OC} ($\mu g/g_{OC}$)
Fresh Water	4.92	4.84	0.061	4.2[a]
Salt Water	4.92	4.84	0.011	0.76[b]

[a]$SQC_{OC} = (10^{4.84}\ L/kg_{OC}) \bullet (10^{-3}\ kg_{OC}/g_{OC}) \bullet (0.061\ ug\ endrin/L) = 4.2\ \mu g\ endrin/g_{OC}$

[b]$SQC_{OC} = (10^{4.84}\ L/kg_{OC}) \bullet (10^{-3}\ kg_{OC}/g_{OC}) \bullet (0.011\ \mu g\ endrin/L) = 0.76\ \mu g\ endrin/g_{OC}$

Since organic carbon is the factor controlling the bioavailability of nonionic organic compounds in sediments, SQC 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 concentration and organic carbon data are available, it is best to convert the sediment concentration to μg chemical/gram organic carbon. These concentrations can then be directly compared to the SQC value. This facilitates comparisons between the SQC and field concentrations relative to identification of hot spots and the degree to which sediment concentrations do or do not exceed SQC values. The conversion from dry weight to organic carbon normalized concentration can be done using the following formula:

$$\mu g\ Chemical/g_{OC} = \mu g\ Chemical/g_{DRY\ WT} \div (\%\ TOC \div 100)$$
$$= \mu g\ Chemical/g_{DRY\ WT} \bullet 100 \div \%\ TOC$$

For example, a freshwater sediment with a concentration of 0.1 μg chemical/$g_{DRY\ WT}$ and 0.5% TOC has an organic carbon-normalized concentration of 20 $\mu g/g_{OC}$ (0.1 $\mu g/g_{DRYWT}$ • 100 ÷ 0.5 = 20 $\mu g/g_{OC}$) which exceeds the SQC of 4.2 $\mu g/g_{OC}$. Another freshwater sediment with the same concentration of endrin (0.1 $\mu g/g_{DRY\ WT}$) but a TOC concentration of 5.0% would have an organic carbon normalized concentration of 2.0 $\mu g/g_{OC}$ (0.1 $\mu g/g_{DRY\ WT}$ • 100 ÷ 5.0 = 2.0 $\mu g/g_{OC}$), which is below the SQC for endrin.

In situations where TOC values for particular sediments are not available, a range of TOC values may be used in a "worst case" or "best case" analysis. In this case, the organic carbon-normalized SQC values (SQC_{OC}) may be "converted" to dry weight-normalized SQC values ($SQC_{DRY\ WT.}$). This "conversion" must be done for each level of TOC of interest:

$$SQC_{DRY\ WT} = SQC_{OC}\,(\mu g/g_{OC}) \bullet (\%\ TOC \div 100)$$

where $SQC_{DRY\ WT}$ is the dry weight normalized SQC value. For example, the SQC value for freshwater sediments with 1% organic carbon is 0.042 μg/g:

$$SQC_{DRY\ WT.} = 4.2\ \mu g/g_{OC} \bullet 1\%\ TOC \div 100 = 0.042\ \mu g/g_{DRY\ WT}$$

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

5.2 UNCERTAINTY ANALYSIS:

Some of the uncertainty of the endrin SQC can be estimated from the degree to which the EqP model, which is the basis for the criteria, can rationalize the available sediment toxicity data. The EqP model asserts that (1) the bioavailability of non-ionic organic chemicals across sediments is equal on an organic carbon basis, and (2) that the effects concentration in sediment ($\mu g/g_{OC}$) can be estimated from the product of the effects concentrations from water only exposures (μg/L) and the partition coefficient K_{OC} (L/kg). The uncertainty associated with the

SQC 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 have been conducted to fulfill the minimum database requirements for the development of SQC (See Section 4.3 and the Technical Basis Document; U.S. EPA, 1993a). These freshwater and saltwater tests span a range of chemicals and organisms; they include exposures using water only and a number of sediments; and they are replicated within each chemical - organisms - exposure media treatment. These data are analyzed using an analysis of variance (ANOVA) to estimate the uncertainty (i.e. the variance) associated with each of these sources of variation: that associated with varying the exposure media; and that associated with experimental error. If the EqP model were perfect, then there would be only experimental error. Therefore, the uncertainty associated with the use of EqP is the variance associated with varying exposure media.

The data used in the uncertainty analysis are illustrated in Figure 4-2. The data for endrin are summarized in Appendix B. LC50s 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 LC50s from water-only exposures ($LC50_W$; μg/L) are related to the organic carbon-normalized LC50s from sediment exposures ($LC50_{s,OC}$; $\mu g/g_{OC}$) via the partitioning equation:

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

The EqP model asserts that the toxicity of sediments expressed on an organic carbon basis equals the toxicity in water tests multiplied by the K_{OC}. Therefore, both $LC50_{S,OC}$ and $K_{OC} \bullet LC50_W$ 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 is tested in water only exposures and using different sediments. Let α represent the random variation due to this source. Also, each experiment is replicated. Let $\in$ represent the random variation due to this source. If the model were perfect, there would be no random variations other than that due to experimental error which is reflected in the replications. Hence α represents the uncertainty due to the approximations inherent in the model and $\in$ represents the experimental error. Let $(\sigma_\alpha)^2$ and $(\sigma_\in)^2$ 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 equation is:

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

where $\ln(LC50)_{i,j,k}$, are either $\ln(LC50_W)$ or $\ln(LC50_{S,OC})$ corresponding to a water only sediment exposure; μ_i are the population 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 $(\sigma_\in)^2$. The maximum likelihood method is used to make these estimates (U.S. EPA, 1993a). The results are shown in Table 5-2.

TABLE 5-2: ANALYSIS OF VARIANCE FOR DERIVATION OF SEDIMENT QUALITY CRITERIA CONFIDENCE LIMITS FOR ENDRIN.

Source of Uncertainty	Parameter ($\mu g/g_{OC}$)	Value
Exposure media	σ_α	0.39
Replication	σ_ϵ	0.21
Sediment Quality Criteria	σ_{SQC}[a]	0.39

[a] $\sigma_{SQC} = \sigma_\alpha$

The last line of Table 5-2 is the uncertainty associated with the SQC; i.e., the variance associated with the exposure media variability.

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

Hence:

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

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

The confidence limits are given in Table 5-3.

TABLE 5-3. SEDIMENT QUALITY CRITERIA CONFIDENCE LIMITS FOR ENDRIN.

Type of Water Body	SQC_{OC} $\mu g/g_{OC}$	Sediment Quality Criteria 95% Confidence Limits ($\mu g/g_{OC}$) Lower	Upper
Fresh Water	4.2	2.0	9.1
Salt Water	0.73	0.35	1.6

The organic carbon normalized SQC is applicable to sediments with an organic carbon fraction of $f_{OC} \geq 0.2\%$. For sediments with $f_{OC} < 0.2\%$, organic carbon normalization does not apply and the sediment quality criteria do not apply.

5.3 COMPARISON OF ENDRIN SQC 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 SQC concentrations and 95% confidence intervals can be inferred using effect concentrations from toxicity tests with benthic species exposed to sediments spiked with endrin and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figures 5-1 and 5-2). This is because effect concentrations in sediments can be predicted from water-only toxicity data and K_{OC} values (See Section 4). Chronically acceptable concentrations are extrapolated from Genus Mean Acute Values (GMAV) from water-only, 96-hour lethality

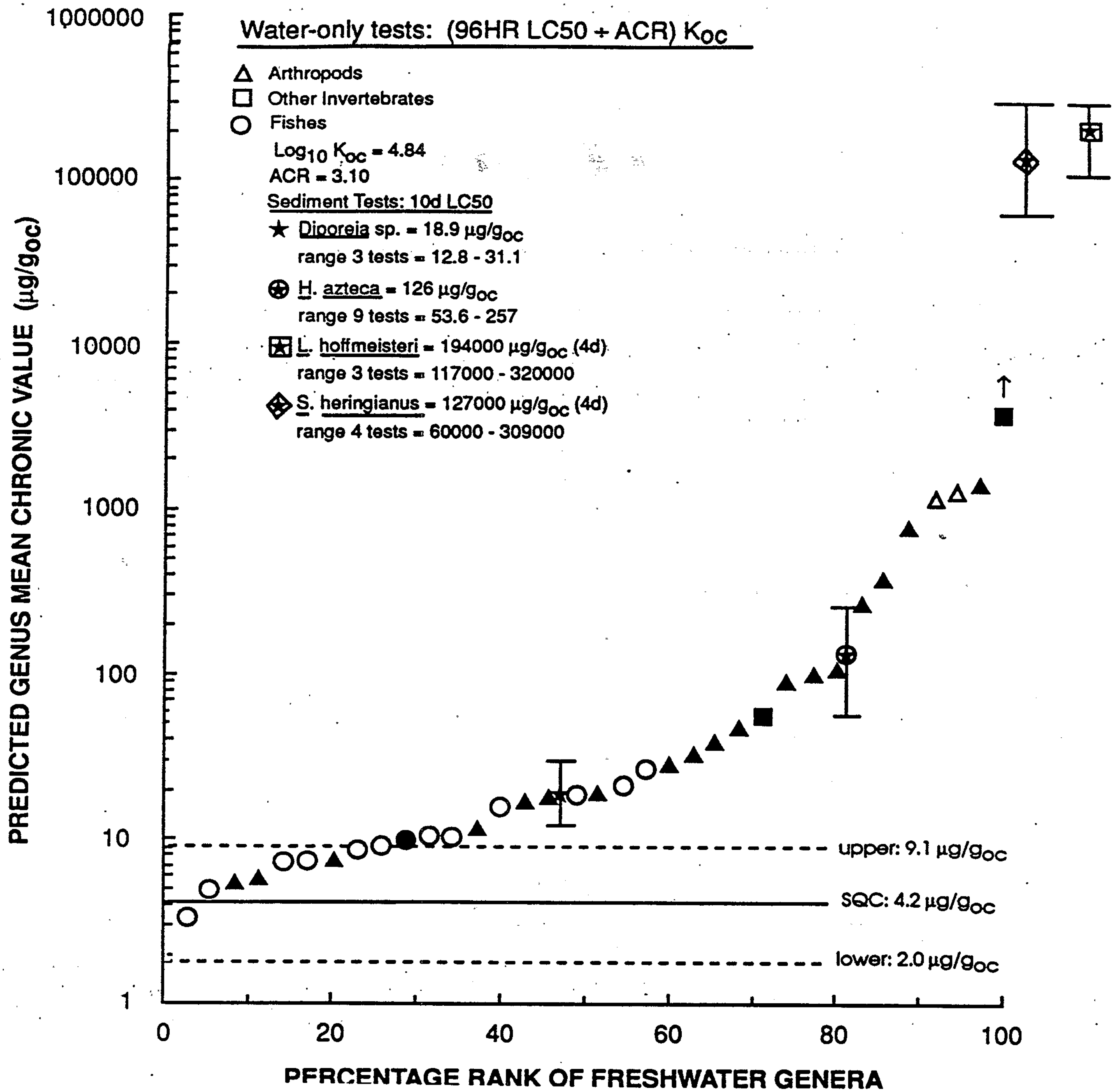


Figure 5-1. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to endrin-spiked sediments and sediment concentrations predicted to be chronically safe in fresh water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV), water-only 96-hour lethality tests, Acute Chronic Ratios (ACR) and K_{oc} values. PGMCV = (GMAV ÷ ACR)K_{oc}. Symbols for PGMCVs are Δ for arthropods, ○ for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

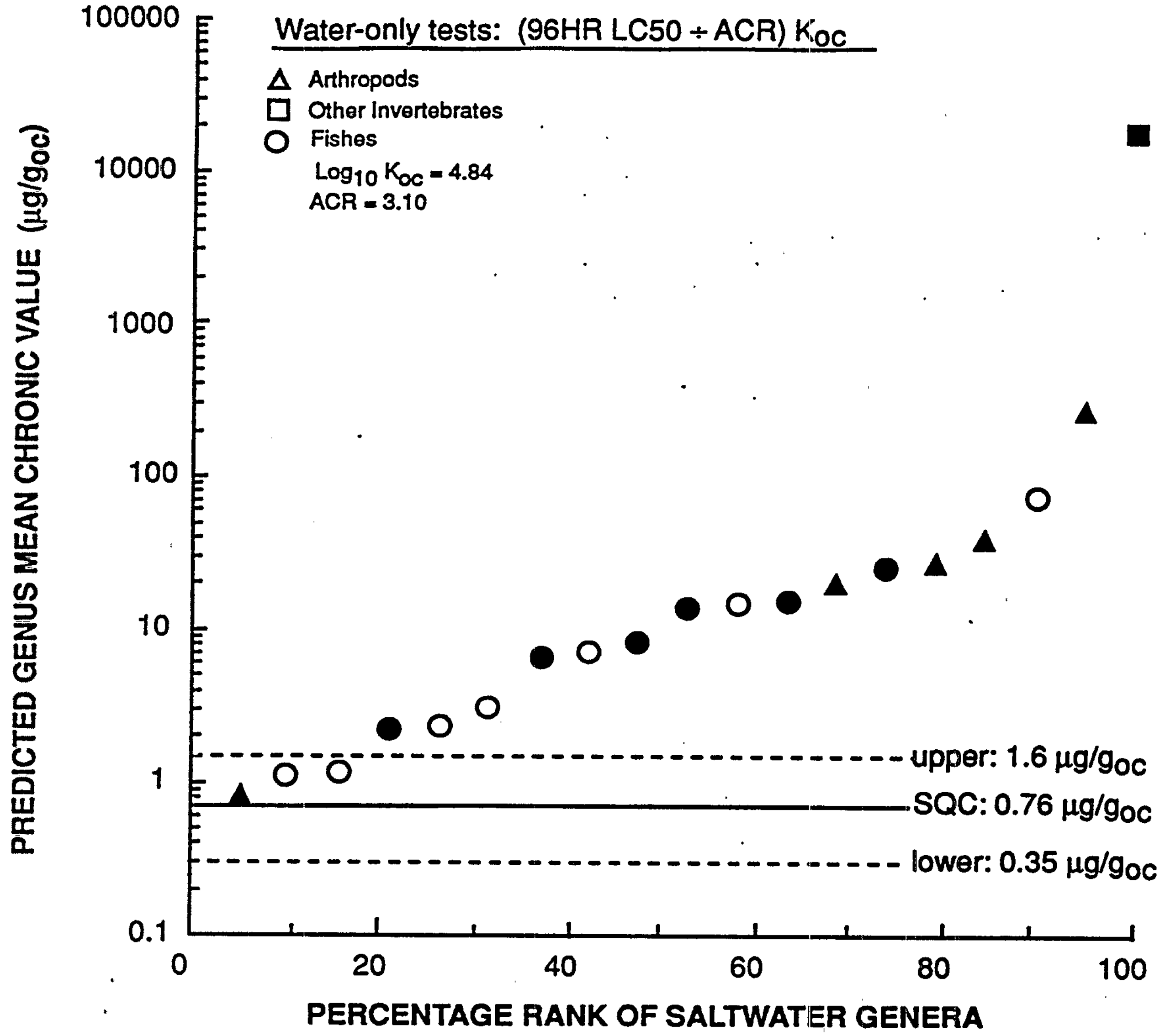


Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to endrin-spiked sediments and sediment concentrations predicted to be chronically safe in salt water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV), water-only 96-hour lethality tests, Acute Chronic Ratios (ACR) and K_{oc} values. PGMCV = (GMAV ÷ ACR)K_{oc}. Symbols for PGMCVs are Δ for arthropods, O for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

tests using acute-chronic ratios (ACR). Therefore, it may be reasonable to combine these two predictive procedures to estimate, for endrin, chronically acceptable sediment concentrations (Predicted Genus Mean Chronic Value (PGMCV)) from GMAVs (Appendix A), ACRs (Table 3-2) and the K_{OC} (Table 5-1):

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

Each predicted GMCV for tested fishes, arthropods or other invertebrates tested in water is plotted against the percentage rank of its sensitivity. Results from toxicity tests with benthic organisms exposed to sediments spiked with endrin (Table 4-1) are placed in the predicted GMCV rank appropriate to the test-specific effect concentration. (For example, the 10-day LC50 for H. azteca, 126 $\mu g/g_{OC}$) is placed between the PGMCV of 105 $\mu g/g_{OC}$ for the mayfly, Ephemerella, and the PGMCV of $\mu g/g_{OC}$ for the dipteran, Tipula.) Therefore, sediment test 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 SQC, organism sensitivities and predicted GMCVs is not intended to provide accurate predictions of the responses of taxa or communities of benthic organisms relative to specific concentrations of endrin in sediments in the field. It is, however, intended to guide scientists and managers through the complexity of available data relative to potential risks to benthic taxa posed by sediments contaminated with endrin.

The freshwater SQC for endrin (4.2 $\mu g/g_{OC}$) is less than 34 of the 35 predicted GMCVs and all of the LC50 values from spiked sediment toxicity tests. The PGMCV for the fish Perca (3 $\mu g/g_{OC}$) is less than the SQC. PGMCVs for 26 of 35 freshwater genera are greater than the upper 95% confidence interval of the SQC (9.1 $\mu g/g_{OC}$). PGMCVs for nine genera, including six water column fish and three benthic arthropod genera are below the SQC 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 impacted by sediment concentrations marginally less than the SQC and possibly less than the 95% upper confidence interval. For endrin, the predicted GMCVs range over three orders of magnitude from the most sensitive to the most tolerant genus. A sediment concentration 10 times the SQC would include the GMCVs of 11 of the 21 benthic genera tested including stoneflies, caddis flies, isopods and fish. Tolerant benthic genera such as the annelid Lumbriculus might be expected to not be chronically impacted in sediments with endrin concentrations 1000X the SQC. Data from lethality tests with two freshwater amphipods, and two freshwater annelids, exposed to endrin spiked into sediments substantiate this range of sensitivity; the LC50s from these tests range from 3 to 80,000 times the SQC of 4.0 $\mu g/g_{OC}$.

The saltwater SQC for endrin (0.76 $\mu g/g_{OC}$) is less than any of the PGMCVs for saltwater genera. The PGMCV for the penaeid shrimp Penaeus (0.83 $\mu g/g_{OC}$) and the fishes Oncorhynchus (1.07 $\mu g/g_{OC}$) and Menidia (1.12 $\mu g/g_{OC}$) are lower than the upper 95% confidence interval for the SQC (1.6 $\mu g/g_{OC}$). For endrin, PGMCVs from the most sensitive to the most tolerant saltwater genus range over four orders of magnitude. A sediment concentration 20 times the SQC would include the GMCVs of one-half of the 12 benthic genera tested including one arthropod and five fish genera. Other genera of benthic arthropods and polychaetes, are less sensitive and might not be expected to be chronically impacted in sediments with endrin concentrations 300X the SQC.

5.4 COMPARISON OF ENDRIN SQC TO STORET AND NATIONAL STATUS AND TRENDS DATA FOR SEDIMENT ENDRIN:

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. Endrin is found at significant concentrations in sediments from rivers, lakes and near coastal water bodies in the United States. This is due to its widespread use and quantity applied during the 1970s and 1980s. It was banned on October 10, 1984. Median concentrations are generally at or near detection limits in most water bodies for data from before and after 1986. There is significant variability in endrin concentrations in sediments throughout the country. Lake samples in EPA Region 9 appear to have relatively high endrin levels (median = 0.030 μg/g) prior to 1986. The upper 10% of the concentrations were disproportionally found in streams, rivers and lakes in EPA Region 7 and streams, rivers, lakes and estuaries in Region 9 prior to 1986. In some streams and rivers in Region 7 concentrations remained high after 1986.

The SQC for endrin can be compared to existing concentrations of endrin in sediments of natural water systems in the United States as contained in the STORET database (U.S. EPA, 1989b). These data are generally reported on a dry weight basis, rather than an organic carbon normalized basis. Therefore, SQC values corresponding to sediment organic carbon levels of 1 to 10% are be compared to endrin's distribution in sediments as examples only. For fresh water sediments, SQC values are 0.042 μg/g dry weight in sediments having 1% organic carbon and 0.42 μg/g dry weight in sediments having 10% organic carbon; for marine sediments SQC are 0.0076 μg/g dry weight and 0.076 μg/g dry weight, respectively. Figure 5-3 presents the

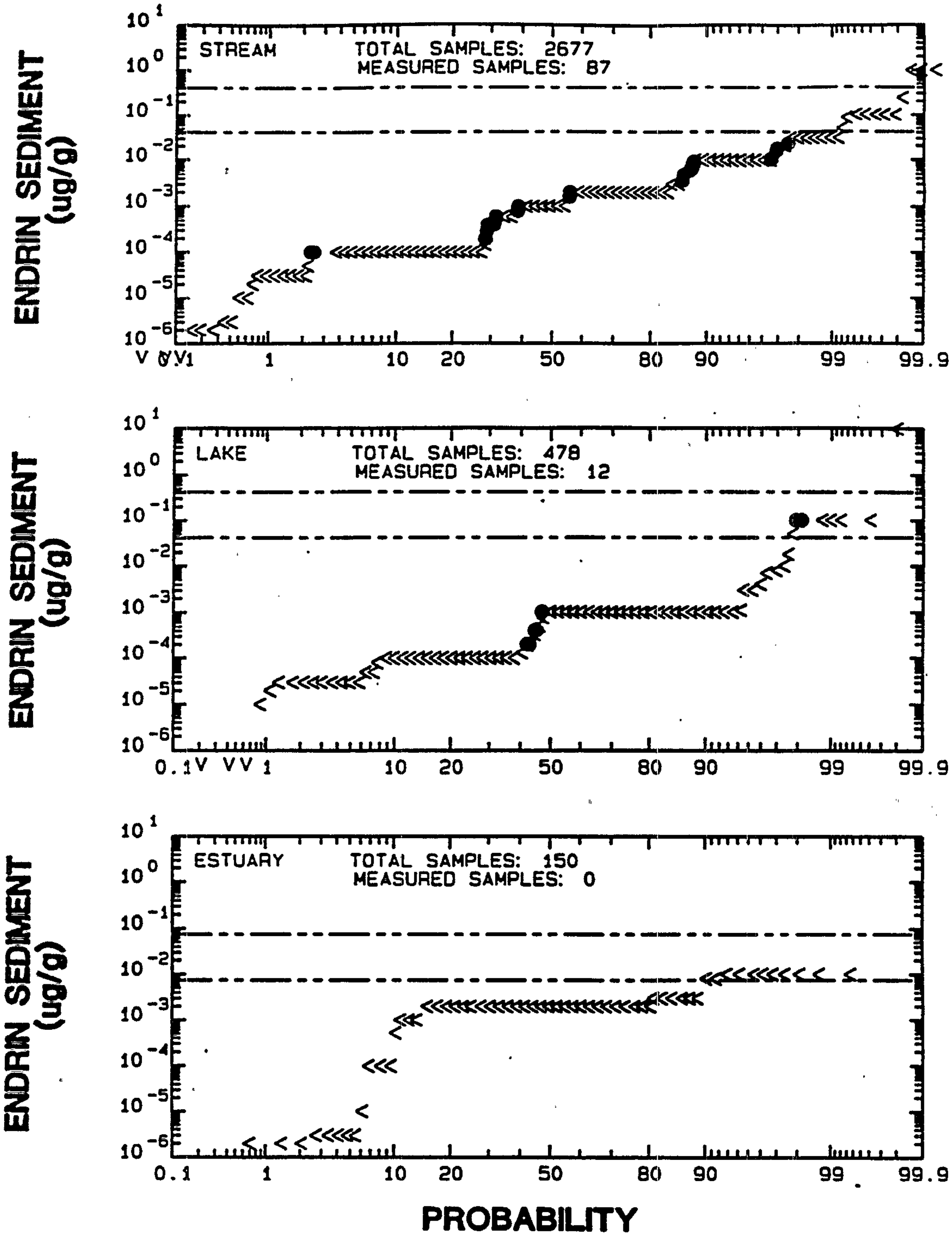


Figure 5-3. Probability distribution of concentrations of endrin in sediments from streams, lakes and estuaries in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared to the endrin SQC values of 0.42 $\mu g/g$ in freshwater sediments having TOC = 10% and 0.042 $\mu g/g$ in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 0.076 $\mu g/g$ when TOC =10% and 0.0076 $\mu g/g$ when TOC=1%. The upper dashed line on each figure represents the SQC value when TOC = 10%, the lower dashed line represents the SQC when TOC = 1%.

comparisons of these SQC to probability distributions of observed sediment endrin levels for streams and lakes (fresh water systems, shown on upper panels) and estuaries (marine systems, lower panel). For streams (n = 2,699) both the SQC of 0.042 μg/g dry weight for 1% organic carbon sediments and the SQC of 0.42 μg/g dry weight criteria for 10% organic carbon freshwater sediments are exceeded by less than 1% of the data. For lakes (n = 478) the SQC for 1% organic carbon sediments is exceeded by about 2% of the data and the SQC of 0.42 μg/g dry weight criteria for 10% organic carbon freshwater sediments is exceeded by less than 1% of the data. For estuaries (n = 150) the SQC of 0.0076 μg/g dry weight for 1% organic carbon salt water sediments are exceeded by about 8% of the data, and the SQC of 0.076 μg/g dry weight criteria for 10% organic carbon freshwater sediments are not exceeded by any of the data. The above description of endrin distributions in Figure 5-3 is misleading because it includes data from most samples in which the endrin concentration was below the detection limit. These data are indicated on the plot as "less than" symbols (<), and plotted at the reported detection limits. Because these values represent upper bounds and not measured values the percentage of samples in which the SQC values are actually exceeded may be less than the percentage reported.

Regional specific differences in endrin concentrations may affect the above conclusions concerning expected criteria 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), the relative frequencies of sampling in different study areas and whether or not the same study areas were sampled during different time periods. It is presented as an aid in assessing the range of reported endrin sediment concentrations and the extent to which they may exceed the SQC.

5.5 LIMITATIONS TO THE APPLICABILITY OF SEDIMENT QUALITY CRITERIA:

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

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 SQC when it occurs with the antagonistic chemical. However, antagonism has rarely been demonstrated. What should be much more common are instances where toxic effects occur at concentrations below the SQC because of the additivity of toxicity of many common contaminants (Alabaster and Lloyd, 1982), e.g. heavy metals and PAHs, and instances where other toxic compounds for which no SQC exist occur along with SQC chemicals.

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

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 each previously at equilibrium. This is particularly relevant in tidal situations where large volumes of sediments are eroded and deposited, yet near equilibrium conditions may predominate over large areas. Except for spills and particulate chemical, near equilibrium is the rule and disequilibrium is uncommon. 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, 1993b).

SECTION 6

CRITERIA STATEMENT

The procedures described in the "Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning" (U.S. EPA, 1993a) indicate that benthic organisms should be acceptably protected in freshwater sediments containing $\leq$ 4.2 μg endrin/g organic carbon and saltwater sediments containing $\leq$ 0.76 μg endrin/g organic carbon, except possibly where a locally important species is very sensitive or sediment organic carbon is < 0.2%.

Confidence limits of 2.0 to 9.1 $\mu g/g_{OC}$ for freshwater sediments and 0.35 to 1.6 $\mu g/g_{OC}$ 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 organic carbon partition coefficient (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, the partitioning and toxicity of endrin, and sound judgement are required in the regulatory use of SQC and their confidence limits.

These concentrations represent the U.S. EPA's best judgement at this time of the levels of endrin in sediments that would be protective of benthic species. It is the philosophy of the Agency and the EPA Science Advisory Board that the use of sediment quality criteria (SQCs) as stand-alone, pass-fail criteria is not recommended for all applications and should frequently trigger additional studies at sites under investigation. The upper confidence limit should be interpreted as a concentration above which impacts on benthic species should be expected. Conversely, the lower confidence limit should be interpreted as a concentration below which impacts on benthic species should be unlikely.

SECTION 7

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Appendix A. - Summary of acute values for endrin for freshwater and saltwater species.

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN- TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
				FRESHWATER SPECIES					
Oligochaete worm, Lumbriculus variegatus	A	I	FT	M	>165	>165	>165	>165	Poirier & Cox, 1991
Cladoceran, Simocephalus serrulatus	X	W,E	S	U	26	-	-	-	Sanders & Cope, 1966
Cladoceran, Simocephalus serrulatus	X	W,E	S	U	45	34	34	34	Sanders & Cope, 1966; Johnson & Finley, 1980
Cladoceran, Daphnia magna	X	W	S	U	352	-	-	-	Gaufin et al., 1965
Cladoceran, Daphnia magna	L	W	S	U	4.2	-	-	-	Johnson & Finley, 1980
Cladoceran, Daphnia magna	L	W	FT	M	230	-	-	-	Thurston et al. 1985
Cladoceran, Daphnia magna	L	W	FT	M	88	142	-	-	Thurston et al. 1985
Cladoceran, Daphnia pulex	L	W	S	U	20	20	53	53	Johnson & Finley, 1980
Ostracod, Cypridopsis vidua	A	I,E	S	U	1.8	1.8	1.8	1.8	Johnson & Finley, 1980
Copepod (cyclopoid), (unidentified)	X	W	S	U	60	60	60	60	Naqvi & Ferguson, 1968
Sowbug, Asellus brevicaudus	A	E	S	U	1.5	1.5	1.5	1.5	Sanders, 1972; Johnson & Finley, 1980
Scud, Gammarus fasciatus	A	E	S	U	4.3	-	-	-	Sanders, 1972; Johnson & Finley, 1980
Scud, Gammarus fasciatus	X	E	S	U	1.3	-	-	-	Sanders, 1972
Scud, Gammarus fasciatus	X	E	FT	U	5.5	3.1	-	-	Sanders, 1972

Appendix A - (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN- TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Scud, Gammarus lacustris	A	E	S	U	3.0	-	-	-	Sanders, 1969; Johnson & Finley, 1980
Scud, Gammarus lacustris	X	E	S	U	11.5	5.9	4.0	4.0	Nebeker & Gaufin, 1964
Glass shrimp, Palaemonetes kadiakensis	A	E	S	U	3.2	-	-	-	Sanders, 1972
Glass shrimp, Palaemonetes kadiakensis	X	E	FT	U	0.5	1.3	1.3	1.3	Sanders, 1972
Crayfish, Orconectes immunis	J	E	FT	M	>89	>89	-	-	Thurston et al. 1985
Crayfish, Orconectes nais	X	E	S	U	320	-	-	-	Sanders, 1972
Crayfish, Orconectes nais	J	E	S	U	3.2	3.2	17	17	Sanders, 1972; Johnson & Finley, 1980
Mayfly, Baetus sp.	J	I	S	U	0.90	0.90	0.90	0.90	Johnson & Finley, 1980
Mayfly, Hexagenia bilineata	X	I	S	U	64	-	-	-	Sanders, 1972
Mayfly, Hexagenia bilineata	J	I	S	U	62	63	63	63	Johnson & Finley, 1980
Mayfly, Ephemerella grandis	X	E	S	U	4.7	4.7	4.7	4.7	Gaufin et al., 1965
Stonefly, Acroneuria pacifica	X	W,E	S	U	0.32	0.32	-	-	Jensen & Gaufin, 1966
Stonefly, Acroneuria sp.	L	W,E	S	U	>0.18	>0.18	0.26	0.26	Johnson & Finley, 1980
Stonefly, Pteronarcella badia	L	I,E	S	U	0.54	0.54	0.54	0.54	Sanders & Cope, 1968; Johnson & Finley, 1980

Appendix A - (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Stonefly, *Pteronarcys californica*	X	I,E	S	U	2.4	-	-	-	Jensen & Gaufin, 1966
Stonefly, *Pteronarcys californica*	A	I,E	S	U	0.25	0.78	0.78	0.78	Sanders & Cope, 1968; Johnson & Finley, 1980
Stonefly, *Claassenia sabulosa*	X	W,E	S	U	0.76	-	-	-	Sanders & Cope, 1968
Stonefly, *Classenia sabulosa*	A	W,E	S	U	0.08	0.25	0.25	0.25	Johnson & Finley, 1980
Caddis fly, *Brachycentrus americanus*	X	E	FT	M	0.34	0.34	0.34	0.34	Anderson & DeFoe, 1980
Damselfly, *Ischnura verticalus*	X	W,E	S	U	1.8	-	-	-	Sanders, 1972
Damselfly, *Ischnura verticalus*	J	W,E	S	U	2.4	2.1	2.1	2.1	Johnson & Finley, 1980
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	Johnson & Finley, 1980
Diptera, *Atherix variegata*	J	I,E	S	U	4.6	4.6	4.6	4.6	Johnson & Finley, 1980
Cutthroat trout, *Oncorhynchus clarki*	J	W	S	U	0.113	-	-	-	Post & Schroeder, 1971
Cutthroat trout, *Oncorhynchus clarki*	J	W	S	U	0.192	0.15	-	-	Post & Schroeder, 1971
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	0.51	-	-	-	Katz, 1961
Coho salmon, *Oncorhynchus kisutch*	J	W	S	U	0.27	-	-	-	Katz & Chadwick, 1961

Appendix A - (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Coho salmon, <u>Oncorhynchus kisutch</u>	J	W	S	U	0.76	0.47	-	-	Post & Schroeder, 1971
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	S	U	0.75	-	-	-	Johnson & Finley, 1980
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	S	U	0.405	-	-	-	Post & Schroeder, 1971
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	S	U	1.1	-	-	-	Macek et al., 1969
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	S	U	0.58	-	-	-	Katz, 1961
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	S	U	0.9	-	-	-	Katz & Chadwick, 1961
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	FT	M	0.33	0.33	-	-	Thurston et al., 1985
Chinook salmon, <u>Oncorhynchus tshawytscha</u>	J	W	S	U	1.2	-	-	-	Katz, 1961
Chinook salmon, <u>Oncorhynchus tshawytscha</u>	J	W	S	U	0.92	1.1	0.40	0.40	Katz & Chadwick, 1961
Brook trout, <u>Salvelinus fontinalis</u>	J	W	S	U	0.355	-	-	-	Post & Schroeder, 1971
Brook trout, <u>Salvelinus fontinalis</u>	J	W	S	U	0.59	0.46	0.46	0.46	Post & Schroeder, 1971
Goldfish, <u>Carassius auratus</u>	J	W	FT	M	0.95	-	-	-	Thurston et al. 1985
Goldfish, <u>Carassius auratus</u>	J	W	FT	U	0.44	-	-	-	Johnson & Finley, 1980
Goldfish, <u>Carassius auratus</u>	J	W	S	U	0.95	0.95	0.95	0.95	Henderson et al., 1959

Appendix A - (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN- TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Carp, *Cyprinus carpio*	J	W	FT	U	0.32	0.32	0.32	0.32	Johnson & Finley, 1980
Bluntnose minnow, *Pimephales notatus*	U	-	FT	U	0.27	-	-	-	Mount, 1962
Bluntnose minnow, *Pimephales notatus*	U	-	FT	U	0.29	-	-	-	Mount, 1962
Bluntnose minnow, *Pimephales notatus*	U	-	FT	U	0.47	0.33	-	-	Mount, 1962
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	-	-	-	Johnson & Finley, 1980
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.26	-	-	-	Solon et al., 1969
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.50	-	-	-	Brungs & Bailey, 1966
Fathead minnow, *Pimephales promelas*	U	-	FT	M	0.49	-	-	-	Brungs & Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.40	-	-	-	Brungs & Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.45	-	-	-	Brungs & Bailey, 1966
Fathead minnow, *Pimephales promelas*	J	W	FT	M	0.64	0.43	0.38	0.38	Thurston et al. 1985

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Black bullhead, *Ictalurus melas*	J	W,E	FT	M	0.45	-	-	-	Anderson & DeFoe, 1980
Black bullhead, *Ictalurus melas*	J	W,E	S	U	1.1	0.45	-	-	Johnson & Finley, 1980
Channel catfish, *Ictalurus punctatus*	J	W,E	S	U	0.32	-	-	-	Johnson & Finley, 1980
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	0.42	0.43	0.43	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	-	-	-	Johnson & Finley, 1980
Mosquitofish, *Gambusia affinis*	X	W	S	U	0.75	-	-	-	Katz & Chadwick, 1961
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	0.9	-	-	-	Katz & Chadwick, 1961
Guppy, *Poecilia reticulata*	X	W	S	U	1.6	1.2	1.2	1.2	Henderson et al., 1959
Threespine stickleback, *Gasterosteus aculeatus*	U	-	S	U	0.44	0.44	0.44	0.44	Katz, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.6	-	-	-	Katz & Chadwick, 1961

Appendix A - (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Bluegill, *Lepomis macrochirus*	U	-	S	U	8.25	-	-	-	Katz & Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	5.5	-	-	-	Katz & Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	2.4	-	-	-	Katz & Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	1.65	-	-	-	Katz & Chadwick, 1961
Bluegill, Lepomis *macrochirus*	J	W	S	U	0.86	-	-	-	Katz & Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.33	-	-	-	Katz & Chadwick, 1961
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.61	-	-	-	Macek et al., 1969; Johnson and Finley, 1980
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.41	-	-	-	Macek et al., 1969
Bluegill, *Lepomis macrochirus*	J	W	S	U	0.37	-	-	-	Macek et al., 1969
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	0.21	0.21	0.21	Thurston et al. 1985
Largemouth bass, *Micropterus salmoides*	J	W	S	U	0.31	0.31	0.31	0.31	Johnson & Finley, 1980

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Yellow perch, <u>Perca flavescens</u>	J	W	FT	U	0.15	0.15	0.15	0.15	Johnson & Finley, 1980
Bullfrog, <u>Rana catesbiana</u>	L	E	FT	M	2.5	2.5	-	-	Thurston et al. 1985
Southern leopard frog, <u>Rana sphenocephala</u>	E	W	FT	M	25	25	2.5(E) 25(W)	7.9	Hall & Swineford, 1980
				SALTWATER SPECIES					
Eastern oyster, <u>Crassostrea virginica</u>	E,L	W,E	S	U	790[j]	790	790	790	Davis & Hidu, 1969
Sand shrimp, <u>Crangon septemspinosa</u>	A	E	S	U	1.7	1.7	1.7	1.7	Eisler, 1969
Hermit crab, <u>Pagurus longicarpus</u>	A	E	S	U	12	12	12	12	Eisler, 1969
Korean shrimp, <u>Palaemon macrodactylus</u>	A	W,E	S	U	4.7	-	-	-	Schoettger, 1970
Korean shrimp, <u>Palaemon macrodactylus</u>	A	W,E	FT	U	0.3	1.2	1.2	1.2	Schoettger, 1970
Grass shrimp, <u>Palaemonetes pugio</u>	L	W	FT	M	1.2	-	-	-	Tyler-Schroeder, 1979
Grass shrimp, <u>Palaemonetes pugio</u>	J	W	FT	M	0.35	-	-	-	Tyler-Schroeder, 1979
Grass shrimp, <u>Palaemonetes pugio</u>	A	W,E	FT	M	0.69	-	-	-	Tyler-Schroeder, 1979
Grass shrimp, <u>Palaemonetes pugio</u>	A	W,E	FT	M	0.63	0.66(E)[k] 0.65(W)	-	-	Schimmel et al., 1975
Grass shrimp, <u>Palaemonetes vulgaris</u>	A	W,E	S	U	1.8	1.8	0.8(W)[k] 0.92(E)	0.8	Eisler, 1969
Pink shrimp, <u>Penaeus duorarum</u>	A	I,E	FT	M	0.037	0.037	0.037	0.037	Schimmel et al., 1975

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
American eel, _Anguilla rostrata_	J	E	S	U	0.6	0.6	0.6	0.6	Eisler, 1970b
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	0.36	0.36	0.36	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	0.95	-	-	Eisler, 1970b
Striped killifish, _Fundulus majalis_	J	W,E	S	U	0.3	0.3	0.65	0.65	Eisler, 1970b
Sailfin molly, _Poecilia latipinna_	A	W	FT	M	0.63	0.63	0.63	0.63	Schimmel et al., 1975
Atlantic silverside, _Menidia menidia_	J	W	S	U	0.05	0.05	0.05	0.05	Eisler, 1970b
Threespine stickleback, _Gasterosteus aculeatus_	J	W,E	S	U	1.65	-	-	-	Katz & Chadwick, 1961
Threespine stickleback, _Gasterosteus aculeatus_	J	W,E	S	U	1.50	-	-	-	Katz & Chadwick, 1961
Threespine stickleback, _Gasterosteus aculeatus_	J	W,E	S	U	1.20	-	-	-	Katz & Chadwick, 1961

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.57	-	-	-	Katz & Chadwick, 1961
Threespine stickleback, *Gasterosteus aculeatus*	J	W,E	S	U	1.57	-	-	-	Katz & 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	1.1	1.1	1.1	Katz, 1961
Striped bass, *Morone saxatilis*	J	E	FT	U	0.094	0.094	0.094	0.094	Korn & Earnest, 1974
Shiner perch, *Cymatogaster aggregata*	J	W	S	U	0.8	-	-	-	Earnest & Benville, 1972
Shiner perch, *Cymatogaster aggregata*	J	W	FT	U	0.12	0.31	0.31	0.31	Earnest & Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	S	U	0.6	-	-	-	Earnest & Benville, 1972
Dwarf perch, *Micrometrus minimus*	A	W	FT	U	0.13	0.28	0.28	0.28	Earnest & Benville, 1972
Bluehead, *Thalassoma bifasciatum*	A	W	S	U	0.1	0.1	0.1	0.1	Eisler, 1970b
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

Appendix A - (continued)

Footnotes:

[a]Lifestage: A = adult, J = juvenile, L = larvae, E = embryo, U = lifestage and habitat unknown, X = lifestage 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, exceptions from Stephan et al (1985).

[f]HMAV species: Habitat Mean Acute Value - Species is the geometric mean of acute values by species for benthic and water column lifestages.

[g]HMAV genus: Geometric mean of HMAV for species within a genus.

[h]Overall GMAV: Geometric mean of acute values across species, habitats and lifestages within the genus.

[i]References: References listed can be found in the Endrin Water Quality Criteria document (U.S. EPA, 1980) or in the references section of this Sediment Quality Criteria document.

[j]Abnormal development of oyster larvae, or loss of equilibrium of brown shrimp or blue crabs.

[k]Habitat mean acute values are listed by habitat when habitats differ between lifestages either within a genus or species.

APPENDIX B. - SUMMARY OF DATA FROM SEDIMENT SPIKING EXPERIMENTS WITH ENDRIN. DATA FROM THESE EXPERIMENTS WERE USED TO CALCULATE K_{oc} VALUES (FIGURE 2-2) AND TO COMPARE MORTALITIES OF AMPHIPODS WITH PORE WATER TOXIC UNITS (FIGURE 4-1) AND PREDICTED SEDIMENT TOXIC UNITS (FIGURE 4-2).

SEDIMENT SOURCE/ SPECIES TESTED	MORTALITY (%)	SEDIMENT CONCENTRATION, µg/g DRY WT.	ORG. CAR.	PORE WATER[a] CONCENTRATION (ug/L)	TOC %	Log K_{oc}[b]	REFERENCES
SOAP CREEK POND	20	2.2	73	1.1	3.0	4.82	Nebeker and
No. 7, OR	32	3.4	113	1.5	3.0	4.88	Schuytema, 1989
	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	9	1.1	18	0.5	6.1	4.56	Nebeker and
SOAP CREEK	44	4.9	80	1.7	6.1	4.67	Schuytema, 1989
POND AND	95	17.7	290	6.8	6.1	4.63	
MERCER LAKE, OR	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 and
	2	1.3	12	0.3	11.2	4.60	Schuytema, 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, OR	1.5	3.0	100	1.1	3.0	4.96	Schuytema et al.,
	8.5	8.7	290	3.1	3.0	4.97	1989
	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 al.,
	5	5.3	48	1.0	11.0	4.68	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	
	100	267	2,430	65.0	11.0	4.57	

APPENDIX B. Endrin (cont'd)

SEDIMENT SOURCE/ SPECIES TESTED	MORTALITY (%)	SEDIMENT CONCENTRATION, μg/g DRY WT.	ORG. CAR.	PORE WATER[a] CONCENTRATION (ug/L)	TOC %	Log K_{oc}[b]	REFERENCES
MERCER LAKE, OR	2.5	1.3	12	0.3	11.0	4.60	Schuytema et 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, 1991
	-	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] "Pore 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 (μ/g_{OC}) + calculated free pore water concentration (μg/L)$\bullet 10^3$ g/kg.