United States
Environmental Protection Agency
Office of Water &
Office of Research and
Development

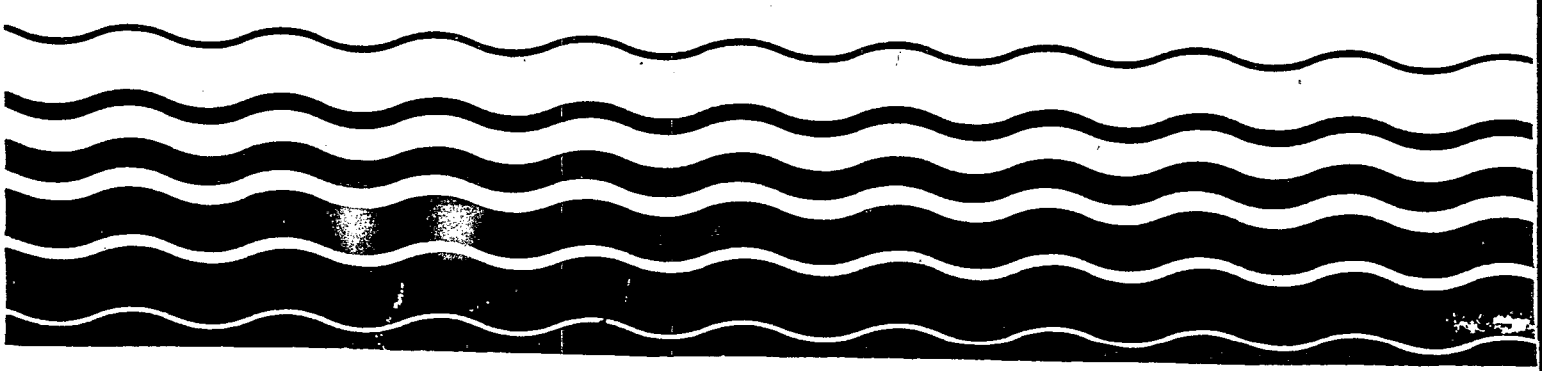
Office of Science and Technology
Health and Ecological Criteria Div.
Washington, D.C. 20460

EPA-822-R-93-015
September 1993

EPA

Sediment Quality Criteria for the Protection of Benthic Organisms:

DIELDRIN



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.

ACKNOWLEDGEMENTS

Principal Author

David J. Hansen	U.S. EPA, Environmental Research Laboratory, Narragansett, RI

Coauthors

Walter J. Berry	Science Applications International Corporation, Narragansett, RI
Dominic M. Di Toro	Manhattan College, Bronx, NY HydroQual, Inc., Mahwah, NJ
Paul R. Paquin	HydroQual, Inc., Mahwah, NJ
Laurie D. De Rosa	HydroQual, Inc., Mahwah, NJ
Frank E. Stancil, Jr.	U.S. EPA, Environmental Research Laboratory, Athens, GA
Robert A. Hoke	Science Applications International Corporation, Hackensack, NJ
Christopher S. Zarba	U.S. EPA Headquarters, Office of Water, Washington, DC

Technical and Clerical Support

Heinz P. Kollig	U.S. EPA, Environmental Research Laboratory, Athens, GA
Glen B. Thursby	Science Applications International Corporation, Narragansett, RI
Maria R. Paruta	NCSC Senior Environmental Employment Program, Narragansett, RI
Dinalyn Spears	Computer Science Corporation, Narragansett, RI
Deborah Robson	Science Applications International Corporation Narragansett, RI

Persons who have made significant contributions to the development of the approach and supporting science used in the derivation of sediment criteria for nonionic organic contaminants are as follows:

Herbert E. Allen	University of Delaware, Newark, DE
Gerald T. Ankley	U.S. EPA, Environmental Research Laboratory, Duluth, MN
Christina E. Cowan	Battelle, Richland, WA
Dominic M. Di Toro	HydroQual, Inc., Mahwah, NJ; Manhattan College, Bronx, NY
David J. Hansen	U.S. EPA, Environmental Research Laboratory, Narragansett, RI
Paul R. Paquin	HydroQual, Inc., Mahwah, NJ
Spyros P. Pavlou	Ebasco Environmental, Bellevue, WA
Richard C. Swartz	U.S. EPA, Environmental Research Laboratory, Newport, OR
Nelson A. Thomas	U.S. EPA, Environmental Research Laboratory, Duluth, MN
Christopher S. Zarba	U.S. EPA Headquarters, Office of Water, Washington, DC

TABLES

FIGURES

($\mu g/L$) • $1Kg_{oc}/1{,}000g_{oc}$). (See Appendix B in this document and Appendix B in the acenaphthene, endrin, 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 dieldrin-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) from 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 dieldrin-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) from 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 dieldrin 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 dieldrin SQC values of 1.1 ug/g in freshwater sediments having TOC = 10% and 0.11 ug/g in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 2.0 ug/g when TOC =10% and 0.20 ug/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%

Figure 5-4. Probability distribution of concentrations of dieldrin in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal dashed line is the saltwater SQC value of 20 $\mu g/g_{oc}$.

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.

AVAILABILITY NOTICE

This document is available to the public through the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161. NTIS Accession Number XXXX-XXXXXX.

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 WQC 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/HEC) 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 nonionic 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 SQC 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 chemical 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, from sediment via ingestion or other sediment-integument exchange or from a mixture of both exposure routes; (4) for nonionic chemicals, effect concentrations in sediments on an organic carbon basis can be predicted using the organic carbon partition coefficient (K_{oc}) and effects concentrations in water; (5) the FCV concentration is an appropriate effects concentration for freely-dissolved chemical in interstitial water; and (6) the 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: (1) pore water is difficult to adequately sample; and (2) significant amounts of the dissolved chemical may be associated with dissolved organic carbon; thus, total chemical concentrations in interstitial water may overestimate exposure.

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 dieldrin 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 to 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 to account for future data.

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, sediment chemical concentrations 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" (US 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 dieldrin. 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 Contaminants" (U.S. EPA, 1993c).

1.2 GENERAL INFORMATION: DIELDRIN

Dieldrin is the common name of a persistent, non-systemic organochlorine insecticide used for control of public health insect pests, termites and locusts. It is formulated for use as an emulsifiable concentrate, wettable and dustable powder and granular product. Other than direct usage of dieldrin, another source of dieldrin in the environment stems from the quick transformation of aldrin, also an organochlorine pesticide, to dieldrin. Both dieldrin and aldrin

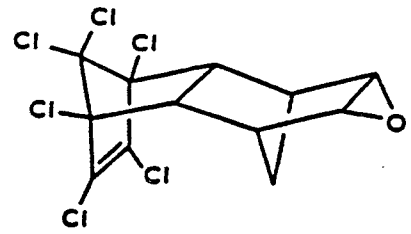
usage peaked in the mid-1960s and declined until the early 1970s. Dieldrin and aldrin have been restricted from registration and production in the United States since 1974 (U.S. EPA, 1980a).

Dieldrin is a cyclic hydrocarbon having a chlorine substituted methanobridge (Figure 1-1). It is structurally similar to endrin, its endo-endo stereoisomer, and has similar physico-chemical chlorine properties, except that it is more difficult to degrade in the environment (Wang, 1988). Dieldrin is a colorless crystalline solid at room temperature, having a melting point of about 176°C and specific gravity of 1.75 at 20°C. It also has a vapor pressure of 0.4 mPa and a solubility of 0.19 mg/L at 20°C (Hartley and Kidd, 1987).

Dieldrin is considered to be toxic to aquatic organisms, bees and mammals (Hartley and Kidd, 1987). The acute toxicity of dieldrin ranges from 0.5 to 740 ug/L for freshwater and 0.7 to >100 μg/L for saltwater organisms (Appendix A). Differences between dieldrin concentrations causing acute lethality and chronic toxicity in species acutely sensitive to this insecticide are small; acute-chronic ratios range from 2.417 to 12.82 for three species (Table 3-3). Dieldrin bioconcentrates in aquatic animals from 400 to 68,000 times the concentration in water (U.S. EPA, 1980a). The WQC for dieldrin (U.S. EPA, 1980a) is derived using a Final Residue Value calculated using bioconcentration data and the FDA action level to protect marketability of fish and shellfish; therefore, the WQC is not "effects based". The SQC for dieldrin is effects based. It is calculated from the Final Chronic Value (FCV) derived in section 3.

1.3 OVERVIEW OF DOCUMENT:

Section 1 provides a brief review of the EqP methodology, and a summary of the



MOLECULAR FORMULA	$C_{12}H_8C1_60$
MOLECULAR WEIGHT	380.93
DENSITY	1.75 g/cc (20°C)
MELTING POINT	176°C
PHYSICAL FORM	Colorless crystal
VAPOR PRESSURE	0.40 mPa (20°C)

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

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

physical-chemical properties and aquatic toxicity of dieldrin. Section 2 reviews a variety of methods and data useful in deriving partition coefficients for dieldrin and includes the K_{oc} recommended for use in the derivation of the dieldrin SQC. Section 3 reviews aquatic toxicity data contained in the dieldrin WQC document (U.S. EPA, 1980a) and new data that were used to derive the FCV used in this document to derive the 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 dieldrin in the derivation of the SQC. Section 4 reviews data on the toxicity of dieldrin in sediments, the need for organic carbon normalization of dieldrin sediment concentrations and the accuracy of the EqP prediction of sediment toxicity using K_{oc} and an effect concentration in water. Data from Sections 2, 3 and 4 are used in Section 5 as the basis for the derivation of the SQC for dieldrin and its uncertainty. The SQC for dieldrin is then compared to STORET (U.S. EPA, 1989b) and National Status and Trends (NOAA, 1991) data on dieldrin's environmental occurrence in sediments. Section 6 concludes with the criteria statement for dieldrin. 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

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 organisms for a wide range of chemicals. (The data showing this for dieldrin are presented in Section 3). Thus, a 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:

$$SQC = K_p \ FCV \qquad (2\text{-}1)$$

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 dieldrin is discussed in this section and 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:

$$K_p = f_{oc} K_{oc} \qquad (2\text{-}2)$$

It follows that:

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

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 dieldrin.

2.2 DETERMINATION OF K_{ow} FOR DIELDRIN:

Several approaches have been used to determine K_{ow} for the derivation of SQC, as discussed in the SQC Technical Basis Document. 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}. The measurement methods were shake-centrifugation (SC), generator column (GCol), and 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 primary references were found listing measured K_{ow}s for dieldrin ranging from 4.09 to 6.2 (Table 2-1). Primary references were found in the literature for estimated $\log_{10}K_{ow}$ ranging from 3.54 to 5.40 (Table 2-1). The range of reported values for dieldrin is significantly greater than the range of values for some other compounds, and we were not able to determine from studying the primary articles that any value was more likely to be accurate than any other.

TABLE 2-1. DIELDRIN MEASURED AND ESTIMATED $LOG_{10}K_{ow}$ VALUES .

METHOD	$LOG_{10}K_{ow}$	REFERENCE
Measured	4.09	Ellington and Stancil, 1988
Measured	4.54	Brooke, et al., 1986
Measured	4.65	De Kock and Lord, 1987
Measured	5.40	De Bruijn et al., 1989
Measured	6.2	Briggs, 1981
Estimated	3.54	Mabey et al., 1982
Estimated	5.40	SPARC[a]

[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 dieldrin 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 dieldrin (Table 2-2). The SC method yielded a $\log_{10}K_{ow} = 5.01$ (n=7), the GCol method yielded a $\log_{10}K_{ow} = 5.16$ (n=7), and the SSF method yielded a $\log_{10}K_{ow} = 5.34$ (n=7). 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 5.34 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 7 K_{ow} measurements made by SSF. The logs of the K_{ow} values measured by SSF range from 5.08 to 5.43.

2.3 DERIVATION OF K_{oc} FROM ADSORPTION STUDIES:

Two types of experimental measurements of the K_{oc} 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 measurements of sediment dieldrin, sediment organic carbon (OC) and non-dissolved organic carbon (DOC)-associated dieldrin dissolved in pore water were used to compute K_{oc}.

2.3.1 K_{oc} FROM PARTICLE SUSPENSION STUDIES:

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

TABLE 2-2. SUMMARY OF $LOG_{10}K_{ow}$ VALUES FOR DIELDRIN MEASURED BY THE U.S. EPA, ENVIRONMENTAL RESEARCH LABORATORY, ATHENS, GA.

SHAKE CENTRIFUGATION (SC)	GENERATOR COLUMN (GCol)	SLOW STIR FLASK (SSF)
5.04	4.89	5.33
5.00	4.88	5.43
5.04	5.18	5.38
5.03	5.15	5.33
5.04	5.26	5.43
4.88	5.38	5.08
4.99	5.67[a]	5.28
	5.04	
5.01[b]	5.16[b]	5.34[b]

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

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

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

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

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

Figure 2-1 compares observed partition coefficient data for the reversible component with 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 observed partition coefficient for dieldrin using adsorption data (Sharom et al., 1980) is highlighted on this plot. The observed $log_{10}Kp$ of 1.68 reflects significant particle interaction effects. The observed partition coefficient is more than an order of magnitude lower than the value expected in the absence of particle effects (i.e., $log_{10}Kp = 3.32$ from the $f_{oc}K_{oc} = 2100$ L/kg). K_{oc} was computed from equation 2-5.

Several sorption isotherm experiments with particle suspensions that provide an additional way to compute K_{oc} were found in a comprehensive literature search for partitioning information for dieldrin (Table 2-3). The K_{oc} values derived from these data are lower than K_{oc} values from laboratory measurements of K_{ow}. The lower K_{oc} can be explained from the particle interaction effects. Partitioning in a quiescent setting would result in less desorption and higher K_{oc}. These data are presented as examples of particle interaction effects only as 100 percent reversibility is assumed in the absence of desorption studies and actual K_{oc} can not be computed.

In the absence of particle effects, K_{oc} is related to K_{ow} via Equation 2-5. For $log_{10}K_{ow}$ = 5.34 (ERL,A, mean measured value), this expression results in an estimate of $log_{10}K_{oc}$ = 5.25.

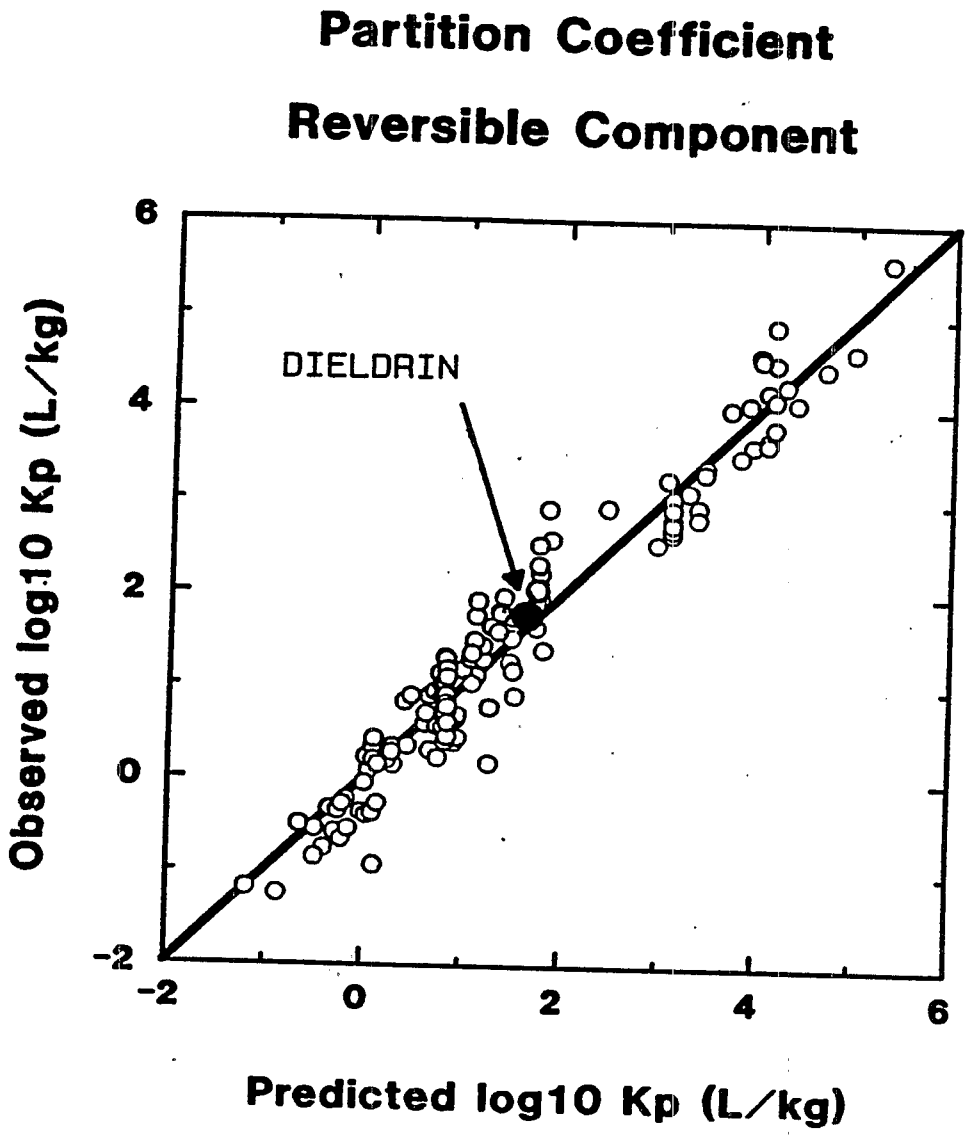


Figure 2-1 Observed versus predicted (equation 2-4) partition coefficients for nonionic organic chemicals (dieldrin datum is highlighted).

2.3.2. K_{OC} FROM SEDIMENT TOXICITY TESTS:

Measurements of K_{OC} are available from sediment toxicity tests using dieldrin (Hoke and Ankley, 1992). These tests were with a sediment having an average organic carbon content of 1.75 percent (Table 4-1; Appendix B). Dieldrin concentrations were measured in sediments and unfiltered pore waters providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. Since it is likely that organic carbon complexing in pore water is significant for dieldrin, organic carbon concentrations were also measured in pore water. Figure 2-2 is a plot of the organic carbon-normalized sorption isotherm for dieldrin, where the sediment dieldrin concentration ($\mu g/g_{OC}$) is plotted versus calculated free (dissolved) pore water concentration ($\mu g/L$). Using pore water organic carbon concentrations (DOC), and assuming K_{DOC} equal to K_{OC}, the calculated free pore water dieldrin concentration C_D ($\mu g/L$) is presented in Figure 2-2 is given by:

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

where C_{PORE} is the measured total pore water concentration and m_{DOC} is the measured DOC concentration (U.S. EPA, 1993a).

The data used to make this plot are included in Appendix B. The line of unity slope corresponding to the $\log_{10}K_{OC} = 5.25$ derived from SSF is compared to the data. The data from the sediment toxicity test fall on the line of unity slope for $\log_{10}K_{OC} = 5.25$.

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 $\text{Log}_{10}\, K_{OC} = 5.32$ and a standard error of the mean of 0.109. This value is in agreement

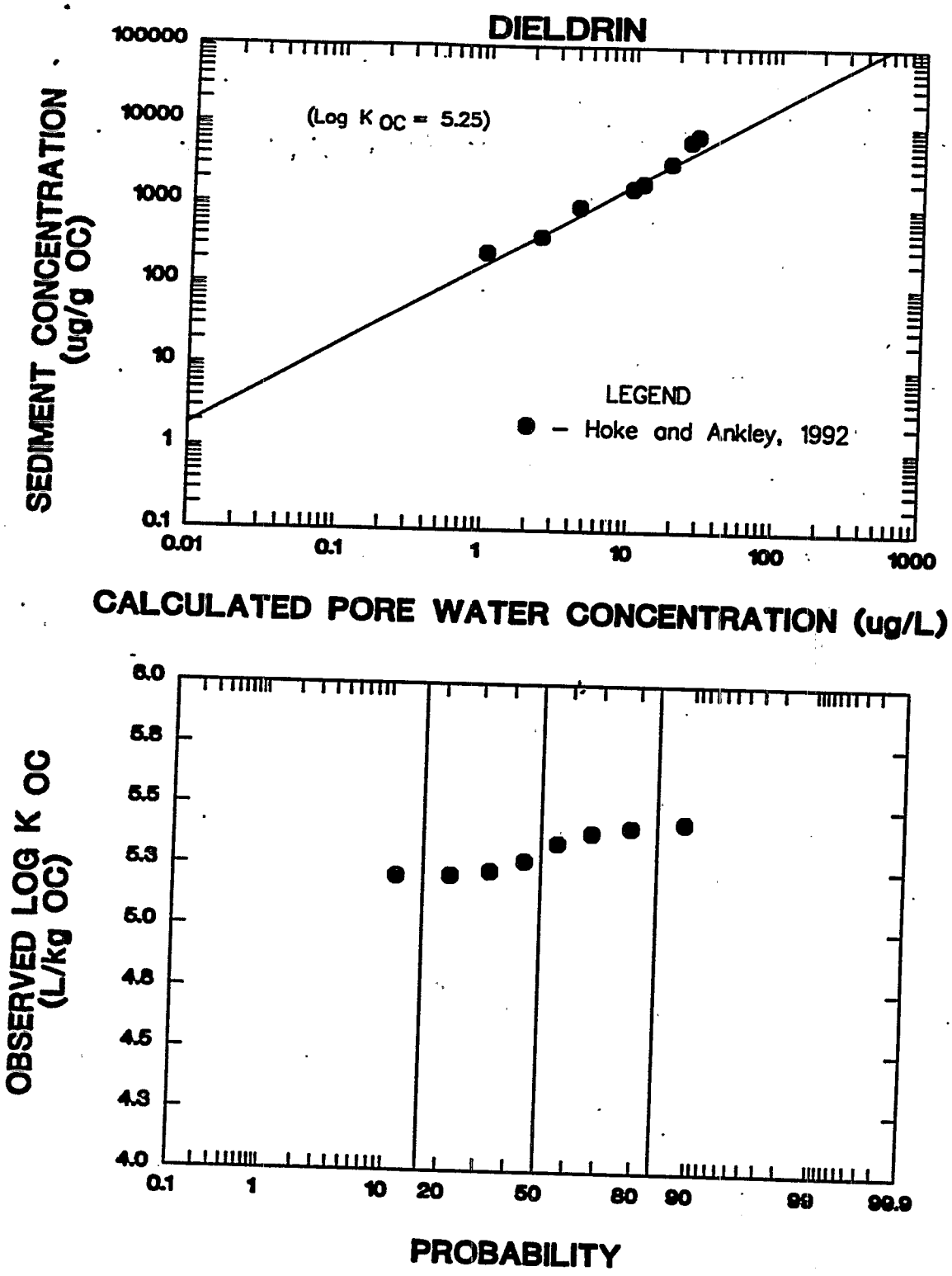


Figure 2-2. Organic carbon-normalized sorption isotherm for dieldrin (top) and probability plot of K_{OC} (bottom) from sediment toxicity tests conducted by Hoke and Ankley (1991). The line in the top panel represents the relationship predicted with a log K_{oc} of 5.25, that is $C_{s,oc}=K_{oc} \bullet C_d$

with $\log_{10}K_{oc}$ = 5.25, which was computed from the SSF determined (Section 2.2) dieldrin $\log_{10}K_{ow}$ of 5.34 (Equation 2-5).

TABLE 2-3. SUMMARY OF K_{oc} VALUES FOR DIELDRIN DERIVED FROM LITERATURE SORPTION ISOTHERM DATA.

Observed $Log_{10}K_{oc}$ (SD)	n	Solids (SD) (g/L)	References
4.20 (0.14)	4	5.0	Eye, 1968
4.14 (0.15)	3	16.4 (4.6)	Betsill, 1990
4.10	1	100.0	Briggs, 1981

2.4 SUMMARY OF DERIVATION OF K_{oc} FOR DIELDRIN:

The K_{oc} selected to calculate the sediment quality criteria for dieldrin is based on the regression of $\log_{10}K_{oc}$ to $\log_{10}K_{ow}$ (Equation 2-5), using the dieldrin $\log_{10}K_{ow}$ of 5.34 recently measured by ERL,A. This approach, rather than the use of the K_{oc} from toxicity tests was adopted because the regression equation is based on the most robust dataset available that spans a broad range of chemicals and particle types, thus encompassing a wide range of K_{ow} and f_{oc}. The regression equation yields a $\log_{10}K_{oc}$ = 5.25. This value is in agreement with the $\log_{10}K_{oc}$ of 5.32 measured in the sediment toxicity tests.

SECTION 3

TOXICITY OF DIELDRIN: WATER EXPOSURES

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

The equilibrium partitioning (EqP) method for derivation of sediment quality criteria (SQC) uses the dieldrin 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 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 dieldrin WQC document (U.S. EPA, 1980a) 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 forty five standard toxicity tests with dieldrin have been conducted on 25 freshwater species from 19 genera (Appendix A). Eighty six of these tests are from one study with the guppy, Poecilla reticulata (Chadwick and Kiigemagi, 1968). Overall genus mean acute values (GMAVs) range from 0.5 to 740 μg/L. Fishes, damselflys, isopods, glass shrimp, stoneflies, and mayflies were most sensitive; GMAVs for these taxa range from 0.5 to 24 μg/L. Seventeen tests on thirteen benthic species from twelve genera are contained in this database (Figure 3-1; Appendix A). Benthic organisms were among both the most sensitive, and most

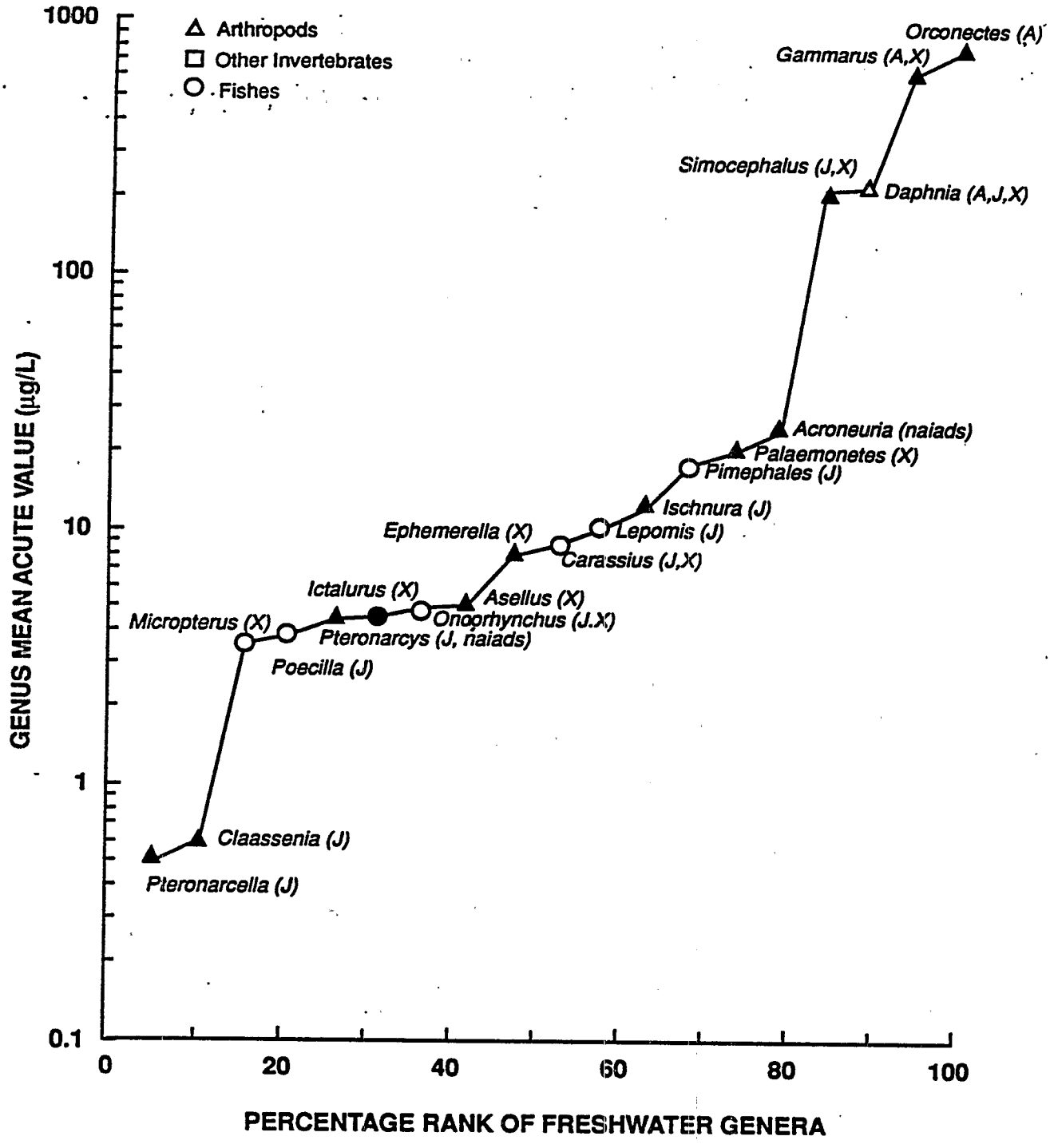


Figure 3-1. Genus mean acute values from water-only acute toxicity tests using freshwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. A = adult, J = juvenile,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

resistant, freshwater species to dieldrin; GMAVs range from 0.5 to 740 μg/L. Of the epibenthic species tested, channel catfish, stoneflies, mayflies, damselflies, and isopods were most sensitive; GMAVs range from 0.5 to 12 μg/L. Infaunal species tested include only the stoneflies, Pteronarcys californica (LC50 = 4.416 μg/L) and Pteronarcella badia (LC50 = 0.5 μg/L). The final acute value (FAV) derived from the overall GMAVs (Stephan et al. 1985) for freshwater organisms is 0.3595 μg/L (Table 3-2).

Thirty two acute tests have been conducted on 23 saltwater species from 21 genera (Appendix A). Overall GMAVs range from 0.70 to > 100 μg/L. Sensitivities of saltwater organisms were similar to those of freshwater organisms. Fishes and crustaceans were the most sensitive. Within this database there are results from 23 tests on benthic life-stages of 16 species from 14 genera (Figure 3-2; Appendix A). Benthic organisms were among both the most sensitive, and most resistant, saltwater genera to dieldrin. The most sensitive benthic species is the pink shrimp, Peneaus duorarum, with a flow-through 96 hour LC50 of 0.70 μg/L based on measured concentrations. The American eel, Anquilla rostrata, has a similar sensitivity to dieldrin with a 96 hr LC50 of 0.9 μg/L. Other benthic species for which there are data appear less sensitive; GMAVs range from 4.5 to > 100 μg/L. The FAV derived from the overall GMAVs (Stephan et al., 1985) for saltwater organisms is 0.6594 μg/L (Table 3-2), less than the acute value for the economically important P. duorarum.

3.3 CHRONIC TOXICITY - WATER EXPOSURES:

Chronic toxicity tests have been conducted with dieldrin using two freshwater fish: rainbow trout, Oncorhynchus mykiss, and the guppy, P. reticulata, and a saltwater mysid,

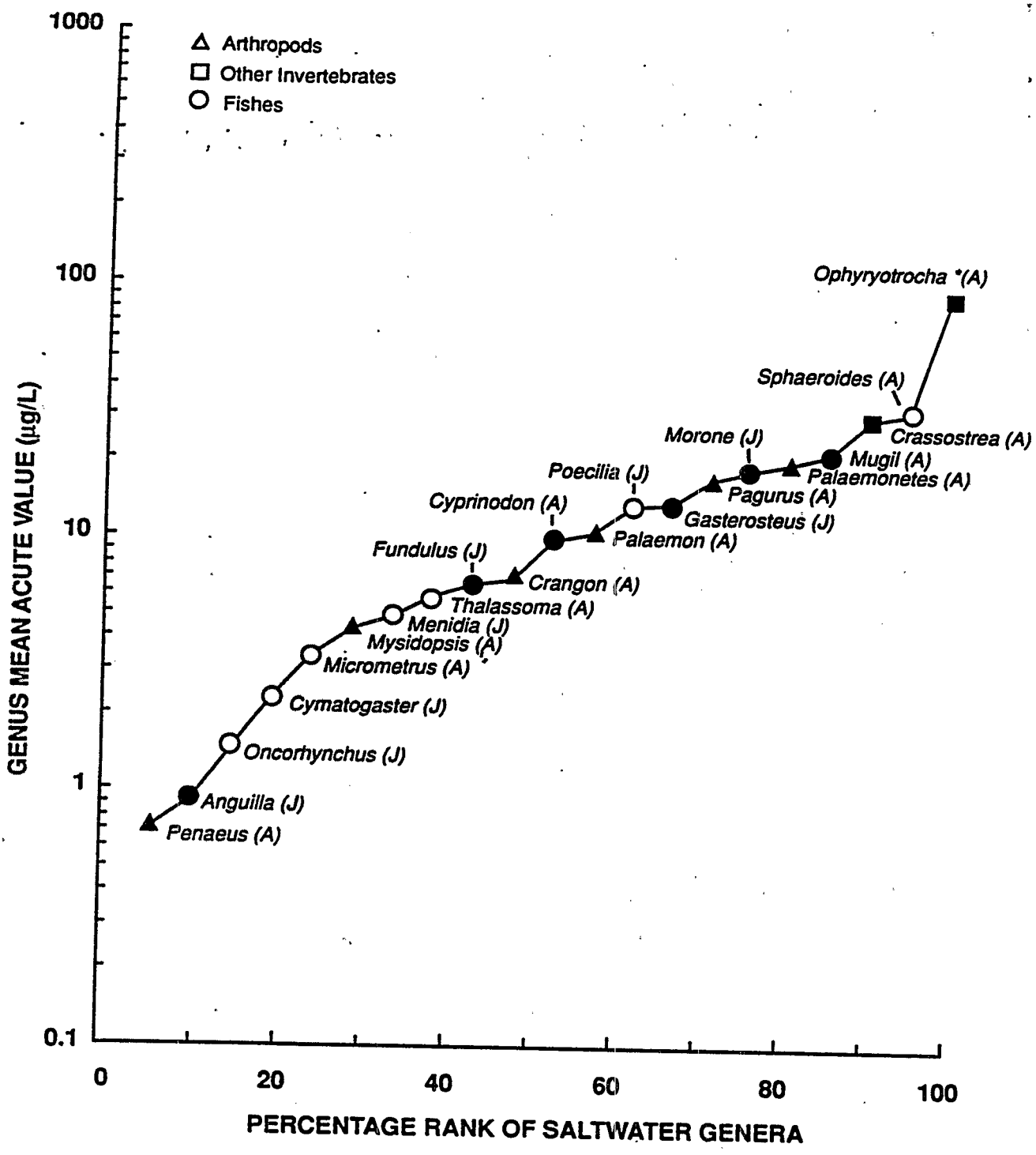


Figure 3-2. Genus mean acute values from water-only acute toxicity tests using saltwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. A = adult, J = juvenile.

Mysidopsis bahia, (Table 3-1). Both O. mykiss and the M. bahia have benthic life stages. (Chronic toxicity tests using O. mykiss and P. reticulata fail to meet the test requirement of measured concentration for use in deriving WQC. Recently, an early life-stage test was successfully completed using rainbow trout, O. mykiss (Brooke, 1993). The acute-chronic ratio ACR, from this test (11.39) was almost identical to the value of 12.82 from unmeasured tests with this fish (Table 3-1; 3-2). This new value will be added to this document following public comment. Time did not permit its inclusion in this draft.

Dieldrin concentrations were not measured in freshwater tests. However, the nominal and measured concentrations in the salt water M. bahia chronic test differed by less than 20% at all concentrations. One life cycle test has been conducted with O. mykiss (Chadwick and Shumway 1969). There was a 97% reduction in survival and a 36% reduction in growth of the survivors in 0.39 μg/L relative to control fish; all fish died at 1.2 μg/L. O. mykiss were not significantly affected at concentrations of 0.012 to 0.12 μg/L. No progeny were tested. The other freshwater chronic test was a three-generation study using the guppy, P. reticulata (Roelofs, 1971). Because exposure concentrations were increased from the test with the first generation to the tests with the next two generations, and because there was no effect at any concentration in the first test, only results from the second two tests are reported here (Table 3-2). There was no effect on P. reticulata survival at dieldrin concentrations from 0.2 to 1.0 μg/L. Mean brood size was reduced by 32% at 2.5 μg/L.

Saltwater M. bahia exposed to dieldrin in a life-cycle test were affected at concentrations similar to those affecting the two freshwater fish mentioned above. M. bahia exposed to 1.1 and 1.6 μg/L (U.S. EPA, 1980b) had a 35% and 58% reduction in survival, respectively, relative

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

Common Name, Scientific Name	Test[a]	Habitat[b] (Lifestage)	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								
Rainbow trout, Oncorhynchus mykiss	ELS	W	0.012-0.12[e]	0.39[e]	97% (M) 36% (G)[f]	-	-	Chadwick & Shumway, 1969
				1.2[e]	100% (M)			
Guppy, Poecilia reticulata[f]	LC	W	0.2-1.0[e]	2.5[e]	32% (R)[h]	2.5	-	Roelofs, 1971
SALTWATER SPECIES								
Mysid, Mysidopsis bahia	LC	E(J,A)	0.10-0.49	1.1	35% (M)	-	-	U.S. EPA, 1980
				1.6	58% (M)			
Polychaete worm, Ophryotrocha diadema	LC	I(L)	0.1[h]	0.3[h]	40% (R)	0.3	35% (M)	Hooftman & Vink, 1980
				1.5	37% (R)	1.5	16% (M)	
				3.1	81% (R)	3.1	61% (M)	
				13	99% (R) 57% (M)	13	71% (M)	
Polychaete worm, Ophryotrocha diadema	PLC	I(A)	1.2	2.6	57% (R)	2.6	39% (M)	Hooftman & Vink, 1980
				8	92% (R)	8	70% (M)	
				23	97% (R)	23	62% (M)	
				72	100% (R) 63% (M)	72	100% (M)	

[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.
[e] Nominal, not measured.
[f] Estimated from graph.
[g] Generations two and three from a three generation study.
[h] Reduction in mean brood size.
[i] Nominal; (less than limit of analytical detection); all other values listed are measured values (there was good agreement between nominal and measured).

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

Common Name, Scientific Name	Acute Value (µg/L)	Chronic Value (µg/L)	Acute-chronic Ratio	Species Mean Acute-Chronic Ratio
		FRESHWATER SPECIES		
Rainbow trout, Oncorhynchus mykiss	2.774	0.2163	12.82	12.82
Guppy Poecilia reticulata	3.822	1.581	2.417	2.417
		SALTWATER SPECIES		
Mysid, Mysidopsis bahia	4.5	0.7342	6.129	6.129
Polychaete worm, Ophryotrocha diadema	>100	0.1732	>577.4	>577.4
Polychaete worm, Ophryotrocha diadema	>100	1.766	>56.63	

Freshwater:

Final Acute Value = 0.3595 µg/L
Final Acute-chronic Ratio = 5.524
Final Chronic Value = 0.0651 µg/L

Saltwater:

Final Acute Value = 0.6594 µg/L
Final Acute-chronic Ratio = 5.748
Final Chronic Value = 0.1194 µg/L

to control M.bahia. There were no significant effects at 0.10 to 0.49 μg/L. No effects were observed on reproduction at any concentration tested and progeny response was not recorded. One life-cycle and one partial life-cycle test were conducted with the polychaete worm, Ophryotrocha diadema (Hooftman and Vink, 1980; Tables 3-1 and 3-2). The observed nominal no effect concentration was of 0.1 μg/L (below limit of analytical detection) for the life-cycle test initiated with larvae and 1.2 μg/L (based on measured concentrations) for the partial life-cycle test initiated with adults. For the life-cycle test with larvae there were 40, 37 81 and 99% decreases in reproductive potential, (combined effect on number of egg masses and embryo survival), relative to carrier control worms at 0.3, 1.5, 3.1 and 13 μg/L, respectively. Embryo survival was reduced by 35, 16, 61 and 71% at dieldrin concentrations of 0.3, 1.5, 3.1 and 13 μg/L, respectively. At 13 μg/L dieldrin survival was reduced to 34% relative to the controls. In the O.diadema partial life-cycle test, reproductive potential was reduced by 57, 92, 97 and 100% relative to the carrier control in concentrations of 2.6, 8, 23 and 72 μg/L. Sixty-three percent of adults in 72 μg/L died. Reductions in egg survival were 39, 70, 62 and 100% relative to controls in concentrations of 2.6, 8, 23 and 72 μg/L, respectively. The chronic sensitivity of this species appears similar to that of the other species tested chronically but acute sensitivity is low: 96 hr LC50 > 100 μg/L for adults and larvae.

The difference between acute and chronic sensitivity to dieldrin for acutely sensitive species is approximately an order-of-magnitude or less (Table 3-2). The acute-chronic ratio (ACR) for acutely insensitive polychaetes was > 56.63 in one test and > 577.4 in a second. The available ACRs for acutely sensitive species are 2.417 for P. reticulata, 6.129 for M. bahia and 12.82 for O. mykiss. The Final Acute-Chronic Ratio (ACR), the geometric mean of these

three values, is 5.748.

The FCVs (Table 3-2), are used as the effect concentrations for calculating the SQC for benthic species. The FCV for freshwater organisms of 0.0625 μg/L is the quotient of the FAV of 0.3595 μg/L and the final ACR of 5.748. Similarly, the FCV for saltwater organisms of 0.1147 μg/L is the quotient of the FAV of 0.6594 μg/L and the final ACR of 5.748.

3.4 APPLICABILITY OF THE WATER QUALITY CRITERION AS THE EFFECTS CONCENTRATION FOR DERIVATION OF THE DIELDRIN 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 benthic and water column 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 species and acute values for the most sensitive water column species for all chemicals; (2) acute values for all benthic species and acute values for all species in the WQC documents across all chemicals after standardizing the LC50 values; (3) FAVs calculated for benthic species alone and FAVs calculated for all species in the WQC documents; and (4) individual chemical comparisons of benthic species vs. all species. Only in this last comparison are dieldrin-specific comparisons of the sensitivity of benthic and all (benthic and water-column) species conducted. The following paragraphs examine the data on the similarity

of sensitivity of benthic and all species for dieldrin.

For dieldrin, benthic species account for 12 out of 19 genera tested in freshwater, and 14 out of 21 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 dieldrin 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 3-1). 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 dieldrin, the test-statistic falls at the 31 percentile of the generated FAV differences. Since the probability is less

than 95%, the hypothesis of no significant difference in sensitivity for freshwater and saltwater species is accepted (Table 3-3).

Since freshwater and saltwater species showed similar sensitivity, a test of difference in sensitivity for benthic and all (benthic and water column species combined, hereafter referred to as "WQC") organisms combining freshwater and saltwater species using the Approximate Randomization method was performed. The test statistic in this case is the difference between the WQC FAV, computed from the WQC LC_{50} values, and the benthic FAV, computed from the benthic organism LC_{50} values. This is slightly different then the previous test for saltwater and freshwater species. The difference is that saltwater and freshwater species in the first test 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 panel of Figure 3-3. The test statistic for this analysis falls at the 72 percentile and the hypothesis of no difference in sensitivity is accepted (Table 3-3). This analysis suggests that the FCV for dieldrin based on data from all tested species is an appropriate effects concentration for benthic organisms.

TABLE 3-3. RESULTS OF APPROXIMATE RANDOMIZATION TEST FOR THE EQUALITY OF THE FRESHWATER AND SALTWATER FAV DISTRIBUTIONS FOR DIELDRIN 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 (19)	Salt (21)	-0.305	31
Benthic vs Water Column + Benthic (WQC)	Benthic (26)	WQC (40)	0.090	72

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

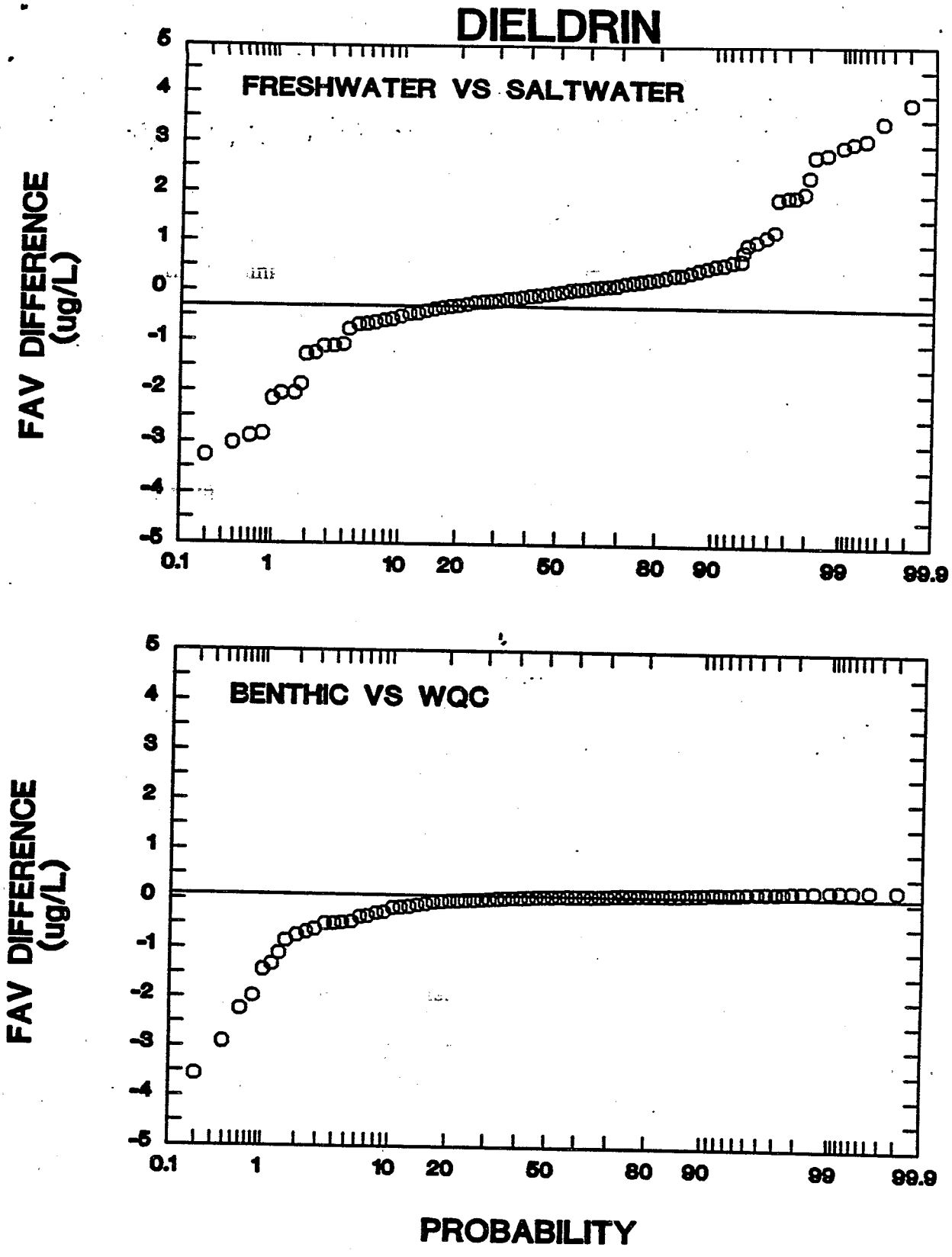


Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater vs. saltwater (upper panel) and benthic vs. WQC (lower panel) data.

SECTION 4

TOXICITY OF DIELDRIN (ACTUAL AND PREDICTED): SEDIMENT EXPOSURE

4.1 TOXICITY OF DIELDRIN IN SEDIMENTS:

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

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

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

Common/Sci. Name	Sediment Source	TOC (%)	Method[a]/ Duration (Days)	Response	Sediment Dieldrin LC50, µg/g Dry wt.	Sediment Dieldrin LC50, µg/g Org. Car.	Pore Water LC50, µg/L	References
			FRESHWATER SPECIES					
Amphipod, Hyalella azteca	Airport Pond, MN	1.7	FT,M/10	LC50	18.2	1,073	57.6	Hoke and Ankley, 1991
Amphipod, Hyalella azteca	West Bearskin Lake, MN	2.9	FT,M/10	LC50	35.0	1,111	220	Hoke and Ankley, 1991
Amphipod, Hyalella azteca	Pequaywan Lake, MN	8.7	FT,M/10	LC50	386	3,682	458	Hoke and Ankley, 1991
Midge, Chironomus tentans	Airport Pond, MN	2.0	FT,M/10	LC50	1.53	78.46	0.50[c]	Hoke, 1992
Midge, Chironomous tentans	Airport Pond, MN	1.5	FT,M/10	LC50	0.53	35.33	0.23[c]	Hoke, 1992
			SALTWATER SPECIES					
Polychaete worm, Nereis virens	17% sand, 83% silt, and clay[b]	2	R,M/12	No mortality	13	650	-	McLeese et al., 1982
Sand shrimp, Crangon septemspinosa	Sand, wet-sieved between 1-2 mm sieves[b]	0.28	R,M/4	LC50	0.0041	1.46	-	McLeese and Metcalfe, 1980

[a]FT = Flow through; M = Measured.
[b]Clean sediment placed in dieldrin-coated beakers at beginning of exposure.
[c]Pore water concentrations estimated from F_{oc}, K_{oc} and measured sediment concentrations.

overlying water varied greatly over the course of these saltwater experiments and exposure conditions are uncertain. In addition, transfer of test organisms to freshly prepared beakers every 48 hours further complicates interpretation of results of McLeese et al. (1982) because exposure conditions change several times during the course of the test. McLeese et al. (1982) tested the effects of dieldrin on the polychaete worm, Nereis virens, in sediment with 2% TOC (17% sand and 83% silt and clay) in 12 day toxicity tests. No worms died in 13 μg/g dry wt sediment, the highest concentration tested. McLeese and Metcalfe (1980) tested the effects of dieldrin in sand with a TOC content of 0.28% on the sand shrimp, Crangon septemspinosa. The 4 day LC50 was 0.0041 ug/g dry wt. sediment, and 1.46 ug/g_{oc}. Concentrations of dieldrin in water overlying the sediment were 10 times the LC50 in water. The authors conclude that sediment-associated dieldrin contributed little towards the toxicity observed.

The effects of dieldrin-spiked sediments from three fresh-water sites in Minnesota on the fresh-water amphipod, Hyalella azteca have been studied by Hoke and Ankley (1991). The total organic carbon (TOC) concentrations in the three sediments were 1.7%, 2.9%, and 8.7%. The sediments were rolled in dieldrin-coated jars at 4°C for 23 days. Mortality of H. azteca in these flow-through tests was related to sediment exposure because dieldrin concentrations in overlying water were generally below detection limits. There was no dose-response relationship observed in the results from the definitive test with one of the sediments (Airport Pond), or in the results from further testing with this sediment using H. azteca (Hoke and Ankley, 1992; Hoke 1992). For this reason only the data from the range finder test with this sediment are used in the analysis of the toxicity data (sections 4.1, 4.2, 4.3), and in Figures 4-1 and 4-2. The ten-day LC50's increased with increasing TOC when dieldrin concentration was expressed on a dry

weight basis, but increased only slightly with increasing organic carbon when dieldrin concentration was expressed on an organic carbon basis (Table 4-1). LC50's normalized to dry weight differed by a factor of 21.2 (18.2 to 386 $\mu g/g$)over a 5.0 fold range of TOC. In contrast, the organic carbon normalized LC50's differed by a factor of 3.4 (1,073 to 3,682 $\mu g/g_{oc}$).

The effects of dieldrin-spiked sediments from two freshwater sites in Minnesota on the fresh water midge, Chironomus tentans, have been studied by Hoke (1992). The TOC contents in the two sediments were 1.5 and 2.0%. The sediments were rolled in dieldrin coated jars at 4°C for one month, stored at 4°C for two months, and then rolled at 4°C for an additional month. LC50s normalized to dry weight differed by a factor of 2.89 (0.53 to 1.53 $\mu g/g$ dry wt). LC50s normalized to organic carbon differed by a factor of 2.22 (35.33 to 78.46). It is not surprising that organic carbon normalization had little effect, given the small range of TOC (1.5 to 2.0%).

Overall, the need for organic normalization of the concentration of nonionic organic chemicals in sediments is presented in the Technical Basis Document (U.S.EPA, 1993a). The need for organic carbon normalization for dieldrin is supported by the dieldrin-spiked toxicity tests described above. Although it is important to demonstrate that organic carbon normalization is necessary if SQC are to be developed using the EqP approach, it is fundamentally more important to demonstrate that K_{oc} and water only effects concentrations can be used to predict the effects concentration for dieldrin and other nonionic organic chemicals on an organic carbon basis for a range of sediments. Evidence supporting this prediction for dieldrin and other nonionic organic chemicals follows in section 4.3.

4.2 CORRELATION BETWEEN ORGANISM RESPONSE AND PORE WATER CONCENTRATION:

One corollary of the EqP theory is that freely dissolved pore-water LC50s for a given organism should be constant across sediments of varying organic carbon content (U.S.EPA, 1993a). Appropriate pore-water values are available from two studies (Table 4-2). Data from tests with water column species were not considered in this analysis. Hoke and Ankley (1991) found 10-day LC50 values for H. azteca based on pore-water concentrations differed by a factor of 8.0 (57.6 to 458 μg/L) for three sediments containing from 1.7 to 8.7% TOC. Therefore, pore water normalized LC50 values provide only a slight improvement over LC50s for dieldrin expressed on a dry weight basis which varied by a factor of 21.2 (18.2 to 386 μg/L). Hoke (1992) found 10-day LC50 values for the C. tentans based on predicted pore water concentrations (the sediment concentration multiplied by the K_{oc}) differed by a factor of 2.17 (0.23 to 0.50). This variability is slightly less than that shown when dry wt (factor of 2.89) is used, but similar to that shown when organic carbon (factor of 2.22) normalization is used. Partitioning to dissolved organic carbon was proposed to explain the lack of similarity of LC50 values based on total pore water dieldrin concentrations.

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), dieldrin (Hoke 1992), endrin (Nebeker et al., 1989; Schuytema et al., 1989), 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

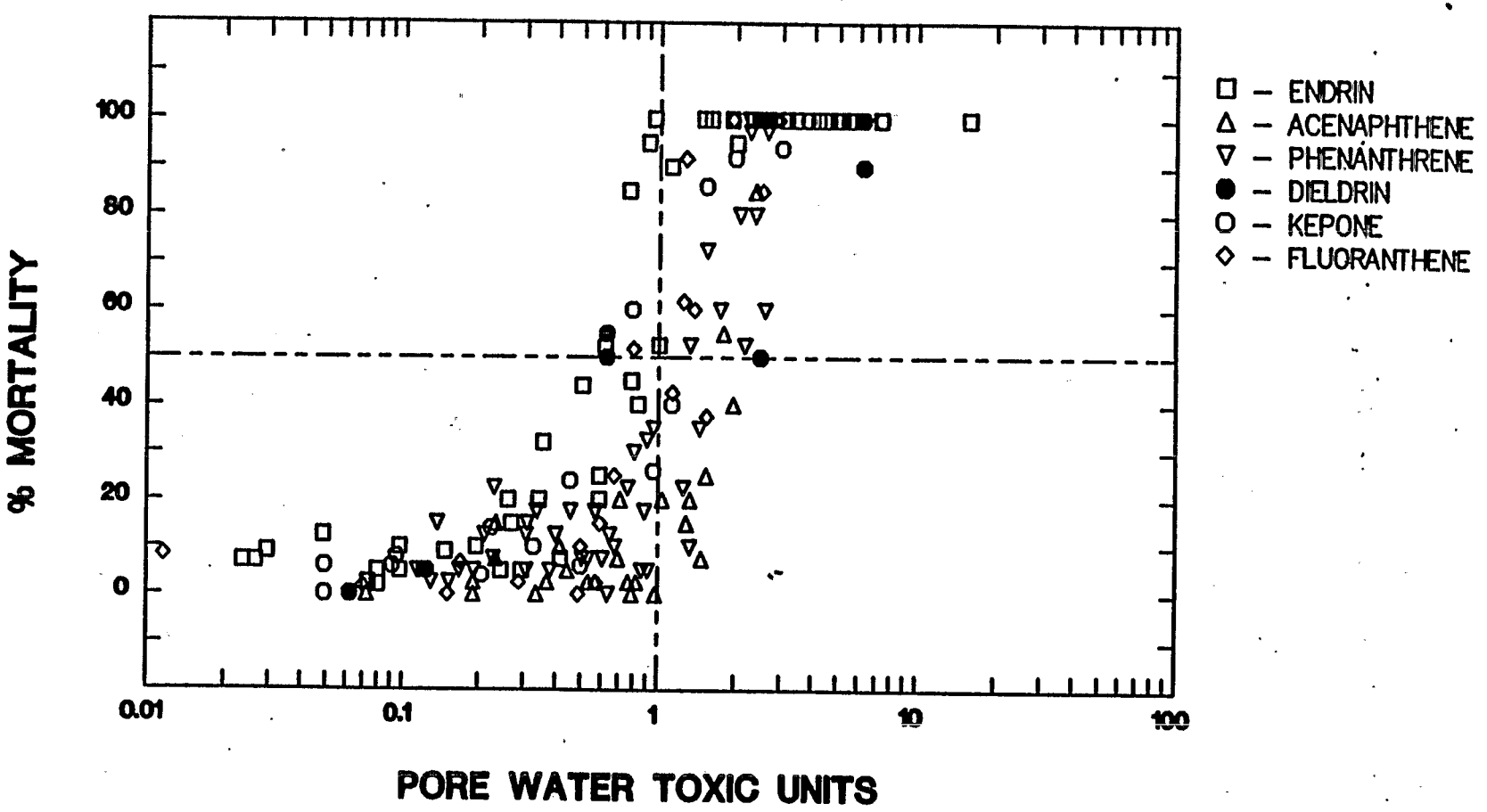


Figure 4-1. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990), and midge in sediments spiked with dieldrin (Hoke, 1992) or kepone (Adams et al., 1985) relative 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 endrin, acenaphthene, fluoranthene and phenanthrene SQC documents, and original references for raw data.)

at the EPA laboratories. Tests with acenaphthene and phenanthrene used two saltwater amphipods (Leptocheirus plumulosus and Eohaustorius estuarius) and marine sediments. Tests with fluoranthene used a saltwater amphipod (Rhepoxynius abronius) and marine sediments. Freshwater sediments spiked with endrin were tested using the amphipod H. azteca; while kepone-spiked sediments were tested using the midge, C. tentans. Figure 4-1 presents the percentage mortalities of the benthic species tested in individual treatments for each chemical versus "pore water toxic units"(PWTU) for all sediments tested. PWTUs are the concentration of the chemical in pore water (μg/L) divided by the water only LC50 (μg/L). Theoretically, 50% mortality should occur at one interstitial 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 pore water normalization was not used to derive SQC in this document because of the complexation of nonionic 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 sediment concentrations and predict biologically available

concentrations across sediment types. The data required to test the organic carbon normalization for dieldrin in sediments are available for 2 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 $\mu g/L$; (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 dieldrin-spiked sediment tests with H. azteca (Hoke and Ankley, 1991) were calculated (Table 4-2) using the $\log_{10} K_{oc}$ value of 5.25 from Section 2 of this document and the water-only LC50 value (7.3 ug/L). Ratios of actual to predicted LC50's for dieldrin averaged 1.26 (range 0.827 to 2.83) in tests with three sediments (Table 4-2). Similarly, predicted 10-day LC50 values for dieldrin-spiked sediment tests with C. tentans were calculated using the $\log_{10} K_{oc}$ of 5.25 and a 10-day water only LC50 value of 0.29 $\mu g/L$. Ratios of predicted to actual LC50s for dieldrin averaged 1.02 (range 0.69 to 1.52) in tests with two sediments (Table 4-2). The overall mean for both species was 1.16.

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 came 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

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

Common/Sci. Name	Method[a] Duration (days)	Water Only LC50 µg/L	Pore Water LC50 µg/L	TOC (%)	Sediment Dieldrin LC50 µg/g Dry Wt.	Sediment Dieldrin LC50 µg/g OC	Predicted[b] LC50 µg/g OC	Ratio: Actual LC50/ Predicted LC50	Reference
			FRESHWATER SPECIES						
Amphipod Hyalella azteca	FT,M/10	7.3	57.6	1.7	18.2	1,073	1,298	0.827	Hoke and Ankley,1991
Amphipod Hyalella azteca	FT,M/10	7.3	220	2.9	35.0	1.111	1,298	0.856	Hoke and Ankley,. 1991
Amphipod Hyalella azteca	FT,M/10	7.3	458	8.7	386	3,682	1,298	2.83	Hoke and Ankley, 1991
Midge, Chironomus tentans	FT,M/10	0.29	0.50[c]	2.0	1.53	78.46	51.6	1.52	Hoke, 1992
Midge, Chironomus tentans	FT,M/10	0.29	0.23[c]	1.5	0.53	35.33	51.6	0.69	Hoke, 1992

[a]FT = Flow through; m = measured.
[b]Predicted LC50 ($\mu g/g_{oc}$) = Water-only LC50 ($\mu g/L$) x K_{oc} (L/Kg_{oc}) x $1Kg_{oc}/1000g_{oc}$; where $K_{oc} = 10^{5.25}$
[c]Pore water concentration estimated from F_{OC}, K_{OC}, and measured sediment concentration.

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 H. azteca to three dieldrin-spiked freshwater sediments having 1.7, 2.9 and 8.7% 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 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 the effects concentration from water only exposures.

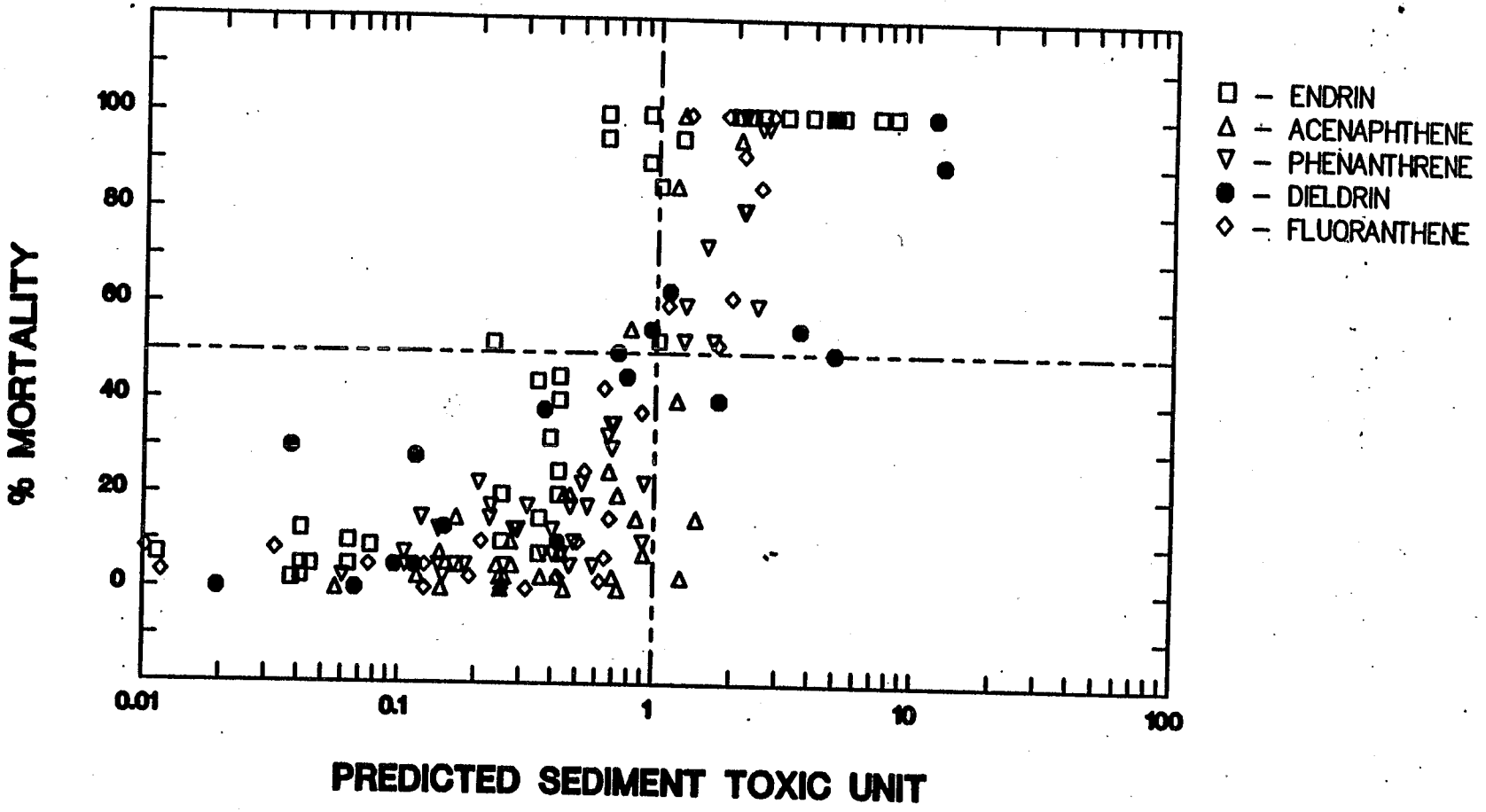


Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke and Ankley, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989) or fluoranthene (Swartz et al., 1990; DeWitt et al., 1992) and midge in dieldrin spiked sediments (Hoke, 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 ($\mu g/L$) • $1Kg_{oc}/1{,}000g_{oc}$). (See Appendix B in this document and Appendix B in the acenaphthene, endrin, fluoranthene, and phenanthrene SQC documents for raw data)

SECTION 5

CRITERIA DERIVATION FOR DIELDRIN

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 to calculate of 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 for the genera from eight or more families 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 dieldrin 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 dieldrin 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 DIELDRIN

Type of Water Body	$Log_{10}K_{ow}$ (L/kg)	$Log_{10}K_{oc}$ (L/kg)	FCV ($\mu g/L$)	SQC_{oc} ($\mu g/g_{oc}$)
Fresh Water	5.34	5.25	0.0625	11[a]
Salt Water	5.34	5.25	0.1147	20[b]

[a] $SQC_{oc} = (10^{5.25}\ L/kg_{oc}) \bullet (10^{-3}\ kg_{oc}/g_{oc}) \bullet (0.0625\ ug\ dieldrin/L) = 11\ \mu g\ dieldrin/g_{oc}$

[b] $SQC_{oc} = (10^{5.25}\ L/kg_{oc}) \bullet (10^{-3}\ kg_{oc}/g_{oc}) \bullet (0.1147\ \mu g\ dieldrin/L) = 20\ \mu g\ dieldrin/g_{oc}$

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 $\mu g/g_{DRYWT} \bullet 100 \div 0.5 = 20\ \mu g/g_{oc}$) which exceeds the freshwater SQC of 11 $\mu g/g_{oc}$. Another freshwater sediment with the same concentration of dieldrin (0.1 $\mu g/g_{DRY\ WT}$) but a TOC concentration of

5.0% would have an organic carbon normalized concentration of 2.0 $\mu g/g_{OC}$ (0.1 $\mu g/g_{DRY\ WT}$ • 100 ÷ 5.0 = 2.0 $\mu g/g_{OC}$), which is below the SQC for dieldrin.

In situations where TOC values for particular sediments are not available, a range of TOC values may be used in a "worst case" or "best case" analysis. In this case, the 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 0.11 $\mu g/g$:

$$SQC_{DRY\ WT.} = 11\ \mu g/g_{OC} \bullet 1\%\ TOC \div 100 = 0.11\ \mu g/g_{DRY\ WT}$$

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

5.2 UNCERTAINTY ANALYSIS:

Some of the uncertainty in the calculation of the dieldrin 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 nonionic organic chemicals from 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 concentration from water only exposures ($\mu g/L$) 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 Technical Basis Document, U.S. EPA, 1993a). These freshwater and saltwater tests span a range of chemicals and organisms; they include both water-only and sediment exposures and they are replicated within each chemical-organism-exposure media treatment. These data were analyzed using an analysis of variance (ANOVA) to estimate the uncertainty (i.e. the variance) associated with varying the exposure media and that associated with experimental error. If the EqP model were perfect, then there would be 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 dieldrin 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 ($LC50_w$; $\mu g/L$) 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 was tested in water-only exposures and in 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 relationship:

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

where $\ln(LC50)_{i,j,k}$, are either $\ln(LC50_w)$ or $\ln(LC50_{s,oc})$ corresponding to a water-only or sediment exposure; μ_i are the population of ln(LC50) for chemical-organism pair i. The error structure is assumed to be lognormal which corresponds to assuming that the errors are proportional to the means, e.g. 20%, rather than absolute quantities, e.g. 1 μg/L. 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.

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

Table 5-2: ANALYSIS OF VARIANCE FOR DERIVATION OF SEDIMENT QUALITY CRITERIA CONFIDENCE LIMITS FOR DIELDRIN.

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

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

The confidence limits for the SQC are computed using this estimate of uncertainty for SQC. 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.

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 do not apply.

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

Type of Water Body	Sediment Quality Criteria 95% Confidence Limits ($\mu g/g_{oc}$) SQC_{oc} $\mu g/g_{oc}$	Lower	Upper
Fresh Water	11	5.2	24
Salt Water	20	9.5	44

5.3 COMPARISON OF DIELDRIN 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 dieldrin and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figures 5-1 and 5-2). 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 value (GMAV) from water-only, 96-hour lethality tests using acute-chronic ratios (ACR). Therefore, it may be reasonable to combine these two predictive procedures to estimate, for dieldrin, 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):

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

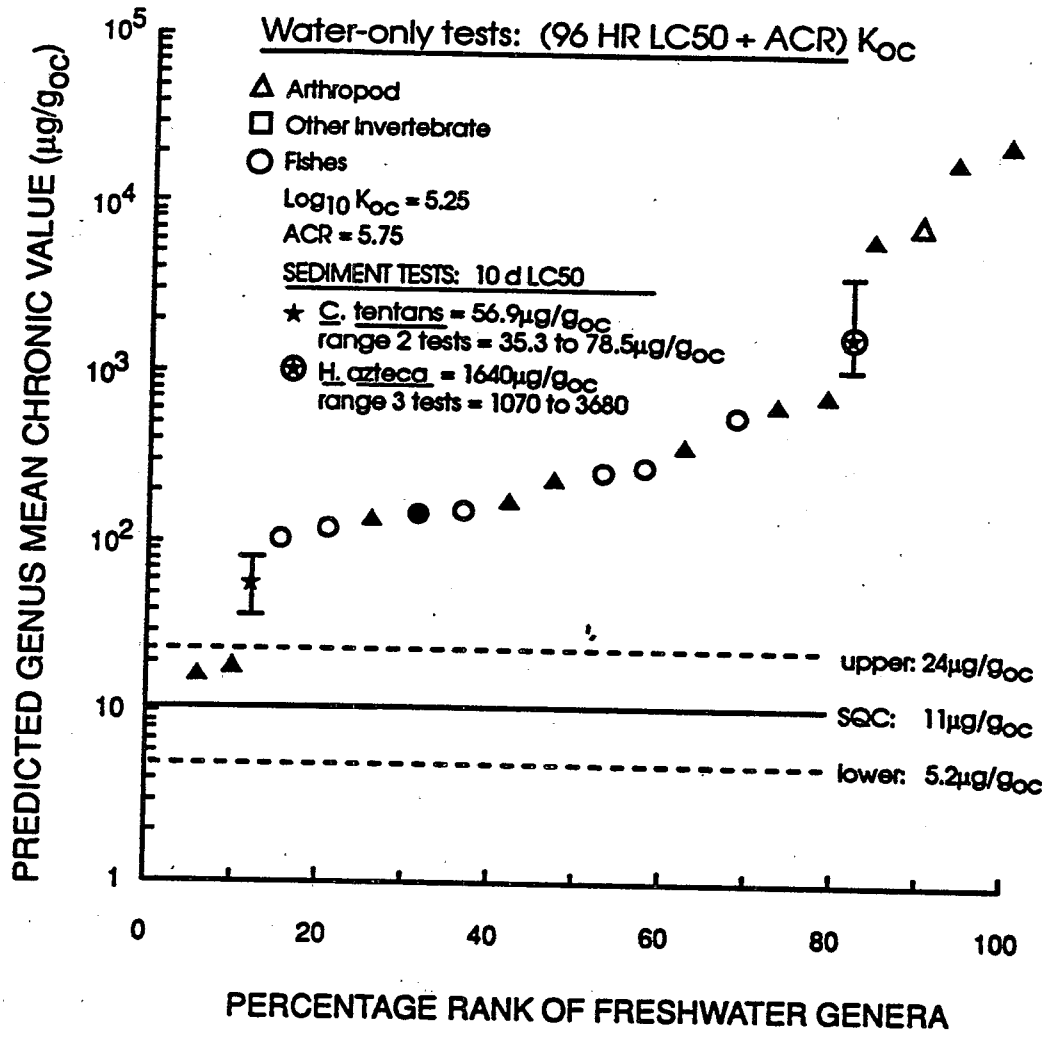


Figure 5-1. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to dieldrin-spiked sediments and sediment concentrations predicted to be chronically safe in fresh water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV) from 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.

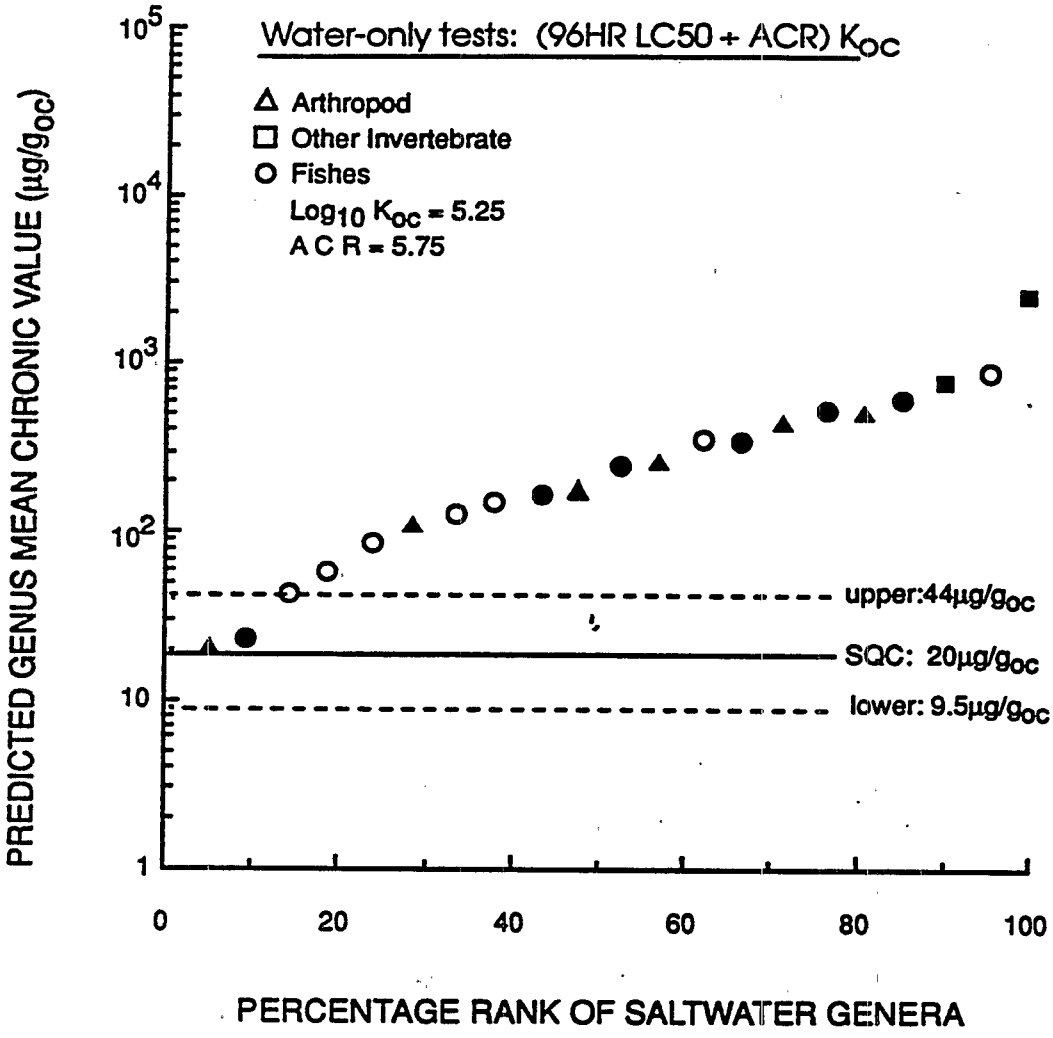


Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to dieldrin-spiked sediments and sediment concentrations predicted to be chronically safe in salt water sediments. Concentrations predicted to be chronically safe (Predicted Genus Mean Chronic Values, PGMCV) are derived from the Genus Mean Acute Values (GMAV) from 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.

In Figures 5-1 and 5-2 each PGMCV for 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 dieldrin (Table 4-1) are placed in the PGMCV rank appropriate to the test-specific effect concentration. (For example, the 10-day LC50 for H. azteca, (1,640 $\mu g/g_{oc}$) is placed between the PGMCV of 742 $\mu g/g_{oc}$ for the stonefly, Acroneuria, and the PGMCV of 6,605 $\mu g/g_{oc}$ for the cladoceran, Simocephalus.) Therefore, 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 PGMCVs is not intended to provide accurate predictions of the responses of taxa or communities of benthic organisms relative to specific concentrations of dieldrin in sediments in the field. It is, however, intended to guide scientists and managers through the complexity of available data relative to potential risks to benthic taxa posed by sediments contaminated with dieldrin.

The freshwater SQC for dieldrin (11 $\mu g/g_{oc}$) is less than any of the PGMCVs or LC50 values from spiked sediment toxicity tests. The PGMCVs for 17 of 19 freshwater genera are greater than the upper 95% confidence interval of the SQC (23 $\mu g/g_{oc}$). The PGMCVs for the stonefly Pteronarcella (15 $\mu g/g_{oc}$) and Claassenia (18 $\mu g/g_{oc}$) 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 above the SQC and possibly

less than the 95% upper confidence interval. For dieldrin, PGMCVs range over three orders of magnitude from the most sensitive to the most tolerant genus. A sediment concentration 20 times the SQC would include the GMCVs of one-half of the 12 benthic genera tested including stoneflies, mayflies, isopods and catfish. Tolerant benthic genera such as the amphipod Gammarus and the crayfish Orconectes might be expected to not be chronically impacted in sediments with dieldrin concentrations 1000X the SQC. Data from lethality tests with another freshwater amphipod, Hyalella, exposed to dieldrin spiked into sediments substantiates this projection; the 10 day LC50s from three tests range from 100 to 350 times the SQC of 11 $\mu g/g_{oc}$.

The saltwater SQC for dieldrin (20 $\mu g/g_{oc}$) is less than PGMCVs any of the PGMCVs for saltwater genera. The PGMCV for the penaeid shrimp Penaeus duorarum (21 $\mu g/g_{oc}$) and the fish Anguilla rostrata (27 $\mu g/g_{oc}$) is lower than the upper 95% confidence interval for the SQC. For dieldrin, PGMCVs from the most sensitive to the most tolerant saltwater genus range over two orders of magnitude. A sediment concentration 16 times the SQC would include the GMCVs of one-half of the 14 benthic genera tested including four arthropod and three fish genera. Other genera of benthic arthropods, polychaetes, molluscs and fishes are less sensitive and might not be expected to be chronically impacted in sediments with dieldrin concentrations 100X the SQC.

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

A STORET (U.S. EPA, 1989b) data retrieval was performed to obtain a preliminary

assessment of the concentrations of dieldrin in the sediments of the nation's water bodies. Log probability plots of dieldrin concentrations on a dry weight basis in sediments are shown in Figure 5-3. Dieldrin is found at varying 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 1960s and early 1970s. It was restricted from registration and production in the United States in 1974. Median concentrations are generally at or near detection limits in most water bodies for data after 1986. There is significant variability with dieldrin concentrations in sediments ranging over nine orders of magnitude within the country.

The SQC for dieldrin can be compared to existing concentrations of dieldrin in sediments of natural water systems in the United States as contained in the STORET database (U.S. EPA, 1989b). These data are generally reported on a dry weight basis, rather than an organic carbon normalized basis. Therefore, SQC values corresponding to sediment organic carbon levels of 1 to 10% are compared to dieldrin's distribution in sediments as examples only. For fresh water sediments, SQC values are 0.11 μg/g dry weight in sediments having 1% organic carbon and 1.1 μg/g dry weight in sediments having 10% organic carbon; for marine sediments SQC are 0.20 μg/g dry weight and 2.0 μg/g, dry weight respectively. Figure 5-3 presents the comparisons of these SQC to probability distributions of observed sediment dieldrin levels for streams and lakes (fresh water systems, shown on the upper panels) and estuaries (marine systems, lower panel). For both streams (n = 3075) and lakes (n = 457), both the SQC of 0.11 μg/g dry weight for 1% organic carbon fresh water sediments and the SQC of 1.1 μg/g dry weight for 10% organic carbon fresh water sediments are exceeded by less than 1% of the data. In estuaries, the data (n=160) indicate that neither criteria, 0.20 ug/g dry weight for sediments

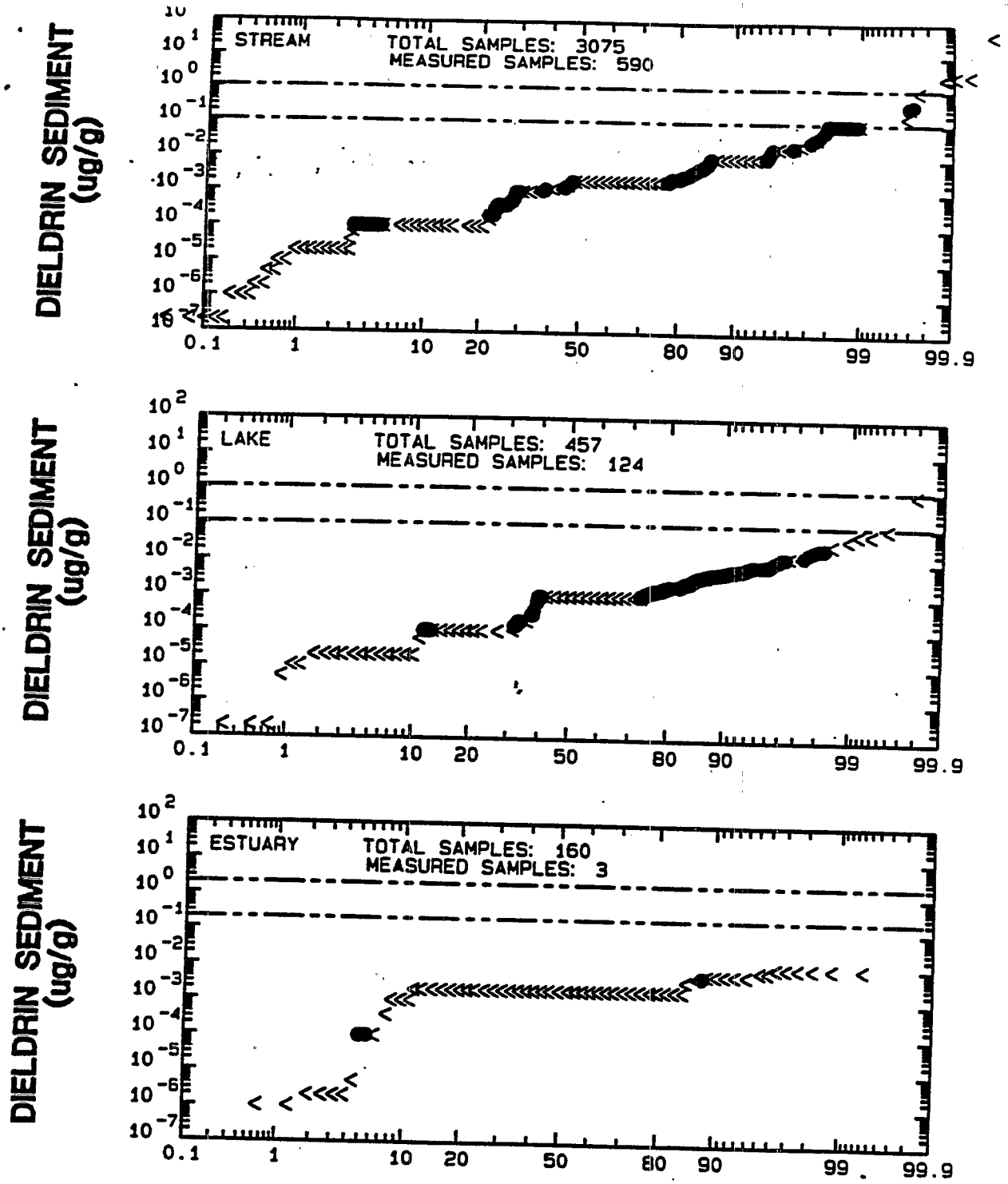


Figure 5-3. Probability distribution of concentrations of dieldrin in sediments from streams, lakes and estuaries in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database compared to the dieldrin SQC values of 1.1 ug/g in freshwater sediments having TOC = 10% and 0.11 ug/g in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 2.0 ug/g when TOC =10% and 0.20 ug/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%

having 1% organic carbon or 2.0 μg/g dry weight for sediments having 10% organic carbon are exceeded by the post 1986 samples. Concentrations of dieldrin in sediments from estuaries are two order of magnitude below the SQC value for 1% organic carbon sediments and three orders of magnitude below the SQC value for sediments with TOCs of 10%.

The dieldrin distribution in Figure 5-3 includes data from some samples in which the dieldrin 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.

A second database developed as part of the National Status and Trends Program (NOAA, 1991) is also available for assessing contaminant levels in marine sediments that are representative of areas away from sources of contamination. The probability distribution for these data, which can be directly expressed on an organic carbon basis, is compared to the saltwater SQC for dieldrin (20 $\mu g/g_{oc}$) on Figure 5-4. Data presented are from sediments with 0.20 to 31.9 percent organic carbon. The median organic carbon normalized dieldrin concentration (0.08 $\mu g/g_{oc}$) is 2 orders of magnitude below the SQC of 20 μg_{oc}. None of these samples (n=408) exceeded the criteria. Hence, these results are consistent with the preceding comparison of the marine SQC to STORET data.

Regional differences in dieldrin 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), or the relative frequencies and intensities of sampling in different study

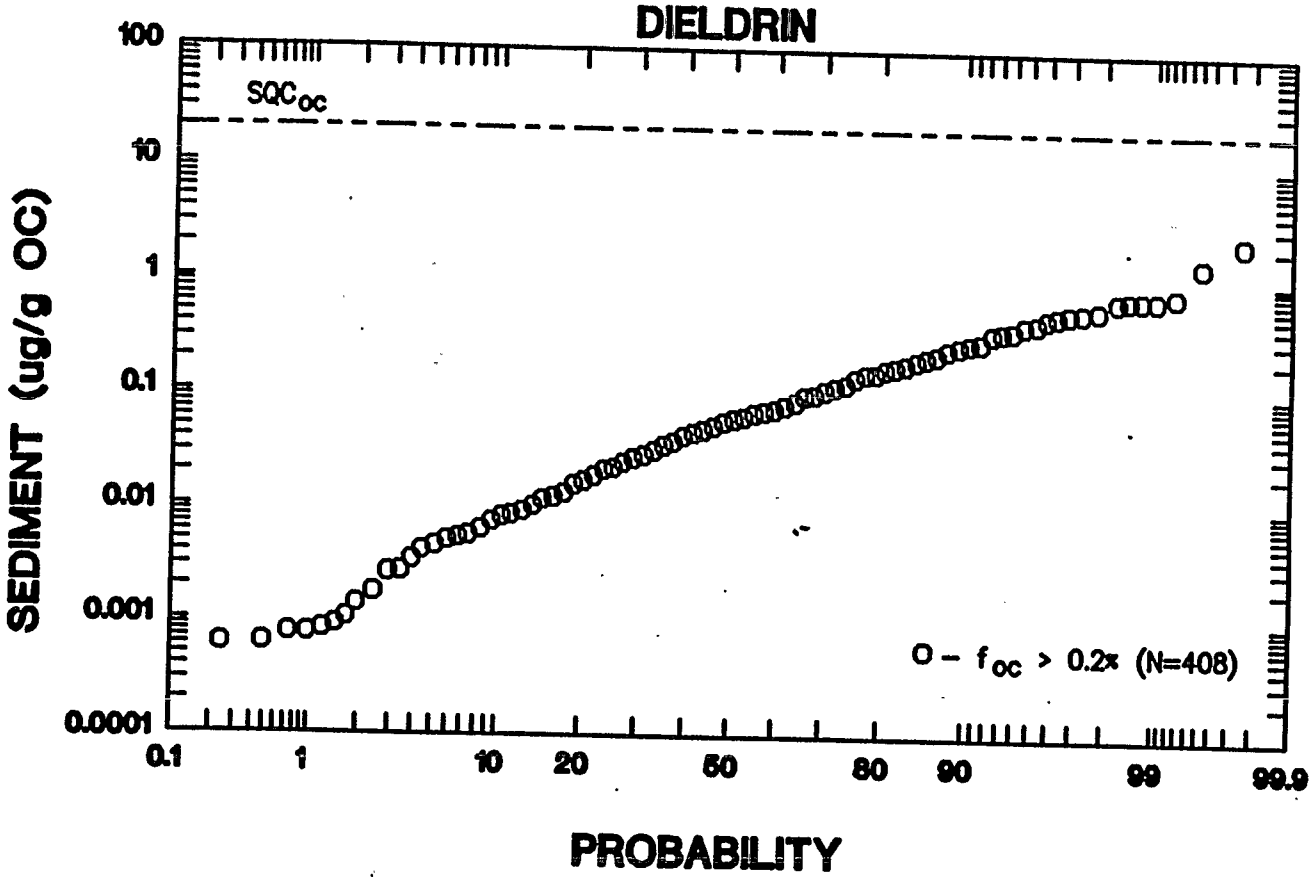


Figure 5-4. Probability distribution of concentrations of dieldrin in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal dashed line is the saltwater SQC value of 20 $\mu g/g_{oc}$.

areas. It is presented as an aid in assessing the range of reported dieldrin sediment concentrations and the extent to which they may exceed the 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 dieldrin 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 National 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 $\leq$ 11 μg dieldrin/g organic carbon and saltwater sediments containing $\leq$ 20 μg dieldrin/g organic carbon, except possibly where a locally important species is very sensitive or sediment organic carbon is < 0.2%.

Confidence limits of 5.2 to 24 $\mu g/g_{oc}$ for freshwater sediments and 9.5 to 44 $\mu g/g_{oc}$ 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 dieldrin, 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 dieldrin 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

REFERENCES

Adams W.J., R.A. Kimerle and R.C. Mosher. 1985. Aquatic safety assessment of chemicals sorbed to sediments. In: Aquatic Toxicology and Hazard Assessment: Seventh Symposium. Eds: R.D. Cardwell, R. Purdy and R.C. Bahner. Amer. Soc. Testing and Materials, Philadelphia, PA. STP 854. pp. 429-453.

Adema, D.M.M. 1978. Dahpnia magna as a test animal in acute and chronic toxicity tests. Hydrobiol. 59:125.

Adema, D.M.M. and G.J. Vink. 1981. A comparative study of the toxicity of 1,1,2-trichloroethane, dieldrin, pentachlorophenol, and 3,4 dichloroaniline for marine and fresh water. Chemosphere 10(6):533-554.

Alabaster, J.S. and R. Lloyd. 1982. Water Quality Criteria for freshwater fish. Chapter 11. Mixtures of Toxicants. London, Butterworth Scientific.

Betsill, J.D. 1990. The sorption of hydrophobic organic compounds in the presence of environmental concentrations of dissolved humic and fulvic acids at variable pH values. Ph.D. Thesis, Oklahoma State University. 155 pp.

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

Brooke, D.N., A.J. Dobbs, and N. Williams. 1986. Octanol: Water partition coefficients (P): Measurements, estimation and interpretation, particularly for chemicals with $P > 10^5$. Ecotoxicol. Environ. Saf. 11:251-260.

Brooke, L.T. 1993. Conducting toxicity tests with freshwater organisms exposed to dieldrin, fluoranthene and phenanthrene. Report to Robert L. Spehar, Project Officer, U.S. EPA. Environmental Research Laboratory-Duluth, Duluth, MN. 18pp.

Butler, P.A. 1963. Commercial Fisheries Investigations. In: Pesticide and Wildlife Studies:

A Review of Fish and Wildlife Service Investigations During 1961 and 1962. U.S. Fish Wildl. Serv. Circ. 167:11.

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

Chadwick, G.G. and U. Kiigemagi. 1968. Toxicity evaluation of a technique for introducing dieldrin into water. Jour. Water Pollut. Control Fed. 40:76-82.

Chadwick, G.G. and D.L. Shumway. 1969. Effects of dieldrin on the growth and development of steelhead trout. In: The Biological Impact of Pesticides in the Environment. Environ. Health Sci. Ser. No. 1, Oregon St. University. p. 90-96.

Chapman, G.A. 1987. Establishing sediment criteria for chemicals-Regulatory perspective. In: Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems. Editors: K.L. Dickson, A.W. Maki and W.A. Brungs. Pergamon Press, New York. p. 355-376.

Chou, J.T. and P.C. Jurs. 1979. Computer-assisted computation of partition coefficients from molecular structures using fragment constants. J. Chem. Inf. Comput. Sci. 19(3):172-178.

Davis, H.C. and H. Hidu. 1969. Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. U.S. Dep. Inter. Fish. Bull. 67:393.

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

De Kock, A.C., and D.A. Lord. 1987. A simple procedure for determining octanol-water partition coefficients using reverse phase high performance liquid chromatography (RPHPLC). Chemosphere 16(1):133-142.

De Witt, T.H., R.J. Ozretich, R.C. Swartz, J.O. Lamberson, D.W. Shults, G.R. Ditsworth, J.K.P. Jones, L. Hoselton, and L.M. Smith. 1992. The effect of organic matter quality on the toxicity and partitioning of sediment-associated fluoranthene to the infaunal marine amphipod, Rhepoxynius abronius. Environmental Toxicology and Chemistry 11:197-208.

Di Toro, D.M. 1985. A particle interaction model of reversible organic chemical sorption. Chemosphere. 14(10):1503-1538.

Di Toro, D.M., C. Zarba, D.J. Hansen, R.C. Swartz, C.E. Cowan, H.E. Allen, N.A. Thomas, P.R. Paquin and W.J. Berry. 1991. Technical basis for establishing sediment quality criteria for nonionic organic chemicals using equilibrium partitioning. Environmental

Toxicology and Chemistry 10:(12)1541-1583.

Douglas, M.T., D.O. Chanter, I.B. Pell, and G.M. Burney. 1986 A proposal for the reduction of animal numbers required for the acute toxicity to fish test (LC50 determination). Aquatic Toxiciology 8(4):243-249.

Earnest, R.D. and P.E. Benville, Jr. 1972. Acute toxicity of four organo-chlorine insecticides to two species of surf perch. Calif. Fish Game. 58:127.

Eisler, R. 1969. Acute toxicities of insecticides to marine decapod crustaceans. Crustaceans. 16:302.

Eisler, R. 1970a. Factors affecting pesticide-induced toxicity in an estuarine fish. U.S. Dept. Interior, Bur. Sport Fish. Wildl. Tech. Paper 45. 18 pp.

Eisler, R. 1979b. Acute toxicities of organic organochlorine and organophosphorous insecticides to estuarine fishes. U.S. Bur. Sport Fish. Wildl., Tech. Pap. 46.

Ellington, J.J., and F.E. Stancil, Jr. 1988. Octanol/water partition coefficients for evaluation of hazardous waste land disposal: Selected chemicals. U.S. EPA, Environmental Research Laboratory, Athens, GA, Environmental Research Brief; EPA/600/M-88/010.

Eye, D. 1967. Aqueous transport of dieldrin in soils. Presented at the 40th Annual Conference of the Water Pollution Control Federation, New York, N.Y.

Gaufin, A.R. et al. 1965. The toxicity of ten organic insecticides to various aquatic invertebrates. Water Sewage Works. 12:276.

Hartley, D. and H. Kidd, eds., 1987. The Agrochemicals Handbook 2nd Editon, Royal Society of Chemistry, University of Nottingham, England.

Henderson, C., et al. 1959. Relative toxicity of ten chlorinated huydrocarbon insecticides to four species of fish. Trans. Am. Fish. Soc. 88:23.

Hoke, R. 1992. Results of the third dieldrin sediment-spiking experiment. Memorandum to D. Hansen, D. Di Toro, and G. Ankley. December 2. 5 pp.

Hoke, R. and G. Ankley, 1991. Results of dieldrin sediment spiking study conducted in support of USEPA development of sediment quality criteria. Memorandum to D. Hansen and D. Di Toro. June 18, 1991. 9 pp.

Hoke, R. and G. Ankley, 1992. Results of re-test of Airport Pond dieldrin-spiked sediments. Memorandum to D. Hansen, D. Di Toro. January 27. 8pp

Hooftman, R.N. and G.J. Vink. 1980. The determination of toxic effects of pollutants with the marine polycheate worm Ophryotrocha diadema. Ecotoxicol. Environ. Safety 4:252-262.

Jensen, L.D. and A.R. Gaufin. 1964. Effects of ten organic insecticides on two species of stonefly naiads. Trans. Am. Fish. Soc. 93:27.

Johnson, W.W. and M.T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. U.S. Department of Interior, Fish and Wildlife Service, Resource Publication 137. 98 pp.

Karickhoff, S.W., L.A. Carreira, C. Melton, V.K. McDaniel, A.N. Vellino, and D.E. Nute. 1989. Computer prediction of chemical reactivity - The ultimate SAR. U.S. EPA, Environmental Research Laboratory, Athens, GA, Environmental Research Brief; EPA/600/M-89/017.

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

Korn, S. and R. Earnest. 1974. Acute toxicity of twenty insecticides to striped bass, Morone sexatilis. Calif. Fish. Game. 60:128.

Mabey, W.R., J.H. Smith, R.T. Podoll, H.L. Johnson, T.Mill, T.W. Chou, J. Gates, I.W. Partridge, H. Jaber, and D. Vandenberg. 1982. Aquatic fate process data for organic priority pollutants. U.S. EPA, Office of Water Regulations and Standards, Washington, DC, Final Report, EPA-440/4-81-014.

Macek, K.J. et al., 1969. The effects of temperature on the susceptibility of bluegills and rainbow trout to selected pesticides. Bull. Environ. Contam. Toxicol. 4:175.

Mackay, D. and B. Powers. 1987. Sorption of hydrophobic chemicals from water: A hypothesis for the mechanism of the particle concentration effect. Chemosphere, 16(4):745-757.

McLeese, D.W., and C.D. Metcalfe. 1980. Toxicities of eight organochlorine compounds in sediment and seawater to Crangon septemspinosa. Bull. Environ. Contam. Toxicol. 25(6):921-928.

McLeese, D.W., L.E. Burridge, and D.J. Dinter. 1982. Toxicities of five organochlorine compounds in water and sediment to Nereis virens. Bull. Environ. Contam. Toxicol. 8(2):216-220.

National Academy of Sciences/National Academy of Engineering. 1973. Water Quality Criteria 1972. EPA.R3.73.033. 594pp.

Nebeker, A.V., G.S. Schuytema, W.L. Griffis, J.A. Barbitta, and L.A. Carey. 1989. Effect of sediment organic carbon on survival of Hyalella azteca exposed to DDT and endrin. Environ.Toxicol. Chem. 8(8):705-718.

NOAA. 1991. National Status and Trends Program - Second summary of data on chemical contaminants in sediments from the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 59. NOAA Office of Oceanography and Marine Assessment, Rockville, MD. 29 pp + appendices.

Noreen, E.W. 1989. Computer intensive methods for testing hypotheses: An introduction. John Wiley and Sons Inc., New York, N.Y.

Parrish, P.R., J.A. Couch, J. Forester, J.M. Patrick Jr., and G.H. Cook. 1973. Dieldrin: Effects on Several Estuarine Organisms. In: Proc. 27th Annu. Conf. S.E. Assoc.Game Fish Comm. p. 427.

Roelofs, T.D. 1971. Effects of dieldrin on the intrinsic rate of increase of the guppy Poecilia reticulata Peters. Ph.D. Thesis, Oregon State University.

Sanders, H.O. 1969. Toxicity of pesticides to the crustacean, Gammarus lacustris. Bur. Sport Fish. Wildl. Tech. Pap. No. 25.

Sanders, H.O. 1972. Toxicity of some insecticides to four species of malacostracan crustaceans. Bur. Sport. Fish. Wildl. Tech. Papa. No. 66.

Sanders, H.O. and O.B. Cope. 1966. Toxicities of several pesticides to two species of cladocerans. Trans. Am. Fish. Soc. 95:165.

Schoettger, R.A. 1970. Fish-Pesticide Research Laboratory, Progress in Sport Fishery Research. U.S. Dept. Int., Bur. Sport Fish Wildl. Res. Publ. 106.

Schuytema, G.A., A.V. Nebeker, W.L. Griffis, and C.E. Miller. 1989. Effects of freezing on toxicity of sediments contaminated with DDT and endrin. Environ. Toxicol. and Chem. 8(10):883-891.

Sharom, M.S., J.R. Miles, C.R. Harris and F.L. McEwen, 1980. Persistence of 12 insecticides in water. Water Research 14:1089-1093.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman and W.A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. PB85-227049. National Technical Information Service, Springfield, VA. 98 pp.

Swartz, R.C. 1991. Acenaphthene and phenanthrene files. Memorandum to David Hansen.

June 26, 1991. 160 pp.

Swartz, R.C., G.R. Ditsworth, D.W. Schults and J.O. Lamberson. 1985. Sediment toxicity to a marine infaunal amphipod: Cadmium and its interaction with sewage sludge. Mar. Environ. Res. 18:133-153.

Swartz, R.C., D.W. Schults, T.H. DeWitt, G.R. Ditsworth and J.O. Lamberson. 1990. Toxicity of fluoranthene in sediment to marine amphipods: A test of the equilibruim partitioning approach to sediment quality criteria. Environ. Toxicol. Chem. 9(8): 1071-1080.

Tarzwell, C.M. and C. Henderson. 1957. Toxicity of dieldrin to fish. Trans. Am. Fish. Soc. 86:245.

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

U.S. Environmental Protection Agency. 1980b. Unpublished laboratory data. Environment Research Laboratory, Gulf Breeze, Florida. 3pp.

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

U.S. Environmental Protection Agency. 1987. Quality Criteria for Water, 1986. EPA 440/5-86-001. May 1, 1987. U.S. Government Printing Office No. 955-002-000008. 406pp.

U.S. Environmental Protection Agency. 1989a. Sediment classification methods compendium. Watershed Protection Division, U.S. EPA. 280 pp.

U.S. Environmental Protection Agency. 1989b. "Handbook: Water Quality Control Information System, STORET," Washington, D.C., 20406.

U.S. Environmental Protection Agency. 1989c. Briefing Report to the EPA Science Advisory Board on the Equilibruium Partitioning Approach to Generating Sediment Quality Criteria. Office of Water Regulations and Standards, Criteria and Standards Division, 132 pp.

U.S. Environmental Protection Agency. 1993a. Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning. (In Review).

U.S. Environmental Protection Agency. 1993b. Guidelines for Deriving Site-Specific

Sediment Quality Criteria for the Protection of Benthic Organisms. (In Review).

U.S. Environmental Protection Agency. 1993c. Guide for the Use and Application of Sediment Quality Criteria for Nonionic Organic Chemicals. (In Review).

Wang, Y.S., 1988. The contamination and bioconcentration of aldrin, dieldrin and endrin in lower lakes at Rocky Mountain Arsenal. Ph.D. Thesis, Colorado State University.

APPENDIX A - SUMMARY OF ACUTE VALUES FOR DIELDRIN FOR FRESHWATER AND SALTWATER SPECIES.

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN- TRATION[d]	EC50	LC50/[e] SPECIES[f]	HMAV GENUS[g]	HMAV GMAV	OVERALL[h] REFERENCES[i]
					µg/L	µg/L	µg/L	µg/L	
FRESHWATER SPECIES									
Cladoceran, Daphnia pulex	J	W	S	U	190	-	-	-	Johnson & Finley, 1980
Cladoceran, Daphnia pulex	X	W	S	U	250	217.9	-	-	Sanders & Cope, 1966
Cladoceran, Daphnia magna	J	W	S	U	>200	-	-	-	Adema, 1978
Cladoceran, Daphnia magna	A	W	S	U	>200	>200	217.9	217.9	Adema, 1978
Cladoceran, Simocephalus serrulatus	J	E,W	S	U	240	-	-	-	Sanders & Cope, 1966; Johnson & Finley, 1980
Cladoceran, Simocephalus serrulatus	X	E,W	S	U	190	213.5	213.5	213.5	Sanders & Cope, 1966
Isopod, Asellus brevicaudus	X	E	S	U	5.0	5.0	5.0	5.0	Sanders, 1972; Johnson & Finley, 1980
Scud, Gammarus fasciatus	A	E	S	U	640	-	-	-	Sanders, 1972; Johnson & Finley, 1980
Scud, Gammarus fasciatus	X	E	S	U	600	619.7	-	-	Sanders, 1972
Scud, Gammarus lacustris	X	E	S	U	700	-	-	-	Gaufin, et al. 1965
Scud, Gammarus lacustris	X	E	S	U	567.5	593.0	593.0	593	Sanders, 1969
Glass shrimp, Palaemonetes kadiakensis	X	E	S	U	20	20	20	20	Sanders, 1972
Crayfish, Orconectes nais	A	E	S	U	740	740	740	740	Sanders, 1972; Johnson & Finley, 1980

Appendix A - Dieldrin: (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b]	ITAT	CONCEN- METHOD[c]	LC50/ TRATION[d]	HMAV EC50	HMAV SPECIES[f]	OVERALL[h]	GENUS[i] GMAVREFERENCES[j]
					µg/L	µg/L	µg/L	µg/L	
Damselfly, Ischnura verticalis	J	E	S	U	12	12	12	12	Johnson & Finley, 1980
Mayfly, Ephemerella grandis	X	E	S	U	7.8	7.8	7.8	7.8	Gaufin et al., 1965
Stonefly, Acroneuria pacifica	(naiads)	E,W	S	U	24.0	24.0	24.0	24.0	Jensen & Gaufin, 1964
Stonefly, Claassenia sabulosa	J	E,W	S	U	0.6	0.6	0.6	0.6	Johnson & Finley, 1980
Stonefly, Petronarcys californica	J	I,E	S	U	0.5	-	-	-	Johnson & Finley, 1980
Stonefly, Pteronarcys californica	(naiads)	I,E	S	U	39.0	4.416	4.416	4.416	Jensen & Gaufin, 1964
Stonefly, Pteronarcella badia	J	I,E	S	U	0.5	0.5	0.5	0.5	Johnson & Finley, 1980
Rainbow trout, Oncorhynchus mykiss	J	W	S	U	9.9	-	-	-	Katz, 1961
Rainbow trout, Oncorhynchus mykiss	J	W	S	U	9.59	-	-	-	Douglas et al., 1986
Rainbow trout, Oncorhynchus mykiss	J	W	S	U	2.4	-	-	-	Macek et al., 1969
Rainbow trout, Oncorhynchus mykiss	J	W	S	U	1.1	-	-	-	Macek et al., 1969
Rainbow trout, Oncorhynchus mykiss	J	W	S	U	1.4	-	-	-	Macek et al., 1969
Rainbow trout, Oncorhynchus mykiss	X	W	S	U	1.2	-	-	-	Johnson & Finley, 1980
Coho salmon, Oncorhynchus kisutch	J	W	S	U	10.8	10.8	-	-	Katz, 1961

Appendix A - Dieldrin: (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		
Chinook salmon, Oncorhynchus tshawytscha	X	W	FT	U	1.54	-	-	-	Schoettger, 1970
Chinook salmon, Oncorhynchus tshawytscha	J	W	S	U	6.1	3.065	-	-	Katz, 1961
Cutthroat trout, Oncorhynchus clarki	X	W	S	U	6.0	6.0	4.845	4.845	Johnson & Finley, 1980
Goldfish, Carassius auratus	J	W	S	U	41	-	-	-	Henderson et al. 1959
Goldfish, Carassius auratus	X	W	S	U	1.8	8.591	8.591	8.591	Johnson & Finley, 1980
Fathead minnow, Pimephales promelas	J	W	S	U	18	-	-	-	Henderson et al. 1959
Fathead minnow, Pimephales promelas	J	W	S	U	18	-	-	-	Henderson et al. 1959
Fathead minnow, Pimephales promelas	J	W	S	U	36	-	-	-	Tarzwell & Henderson, 1957
Fathead minnow, Pimephales promelas	J	W	S	U	24	-	-	-	Tarzwell & Henderson, 1957
Fathead minnow, Pimephales promelas	J	W	S	U	16	-	-	-	Tarzwell & Henderson, 1957
Fathead minnow, Pimephales promelas	J	W	S	U	25	-	-	-	Tarzwell & Henderson, 1957
Fathead minnow, Pimephales promelas	J	W	S	U	23	-	-	-	Tarzwell & Henderson, 1957
Fathead minnow, Pimephales promelas	J	W	S	U	3.8	17.73	17.73	17.73	Johnson & Finley, 1980
Guppy, Poecilia reticulata	J	W	S	U	0.2700	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	0.2128	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.3625	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.7052	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.5811	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.6273	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.7238	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.440	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.356	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.904	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.920	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.640	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.808	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.376	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.304	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	3.219	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.912	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.306	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.328	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.496	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.047	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.430	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.047	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.5590	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.7920	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.320	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.616	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.565	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.940	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	3.475	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.726	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.797	-	-	-	Chadwick & kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.884	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.672	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.645	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	6.048	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	7.869	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.000	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.666	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.290	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.262	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.754	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.976	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	4.165	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.483	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.192	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.840	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.403	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.332	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.277	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.480	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	5.324	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	5.280	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	7.524	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	7.458	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	6.552	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	6.893	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	6.975	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	9.100	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	5.940	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.818	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	5.865	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.713	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	6.375	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.7138	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.6992	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.036	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.284	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.661	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.960	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.032	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	4.818	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	5.865	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.713	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	6.375	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.7138	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	0.6992	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.036	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.284	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.661	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	1.960	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.032	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.625	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.100	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.072	-	-	-	Chadwick & Kiigemagi, 1968

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	3.225	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.165	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.563	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.181	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.488	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.173	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	4.032	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.569	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.010	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	3.280	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	J	W	S	U	2.660	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	X	W	S	U	3.431	-	-	-	Chadwick & Kiigemagi, 1968
Guppy, Poecilia reticulata	X	W	S	U	25	-	-	-	Henderson et al. 1959
Guppy, Poecilia reticulata	X	W	S	U	21	-	-	-	Cairns & Loos, 1966

Appendix A - Dieldrin: (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		
Guppy, Poecilia reticulata	J	W	S	U	3.2	-	-	-	Adema & Vink, 1981
Guppy, Poecilia reticulata	J	W	S	U	7	3.822	3.822	3.822	Adema & Vink, 1981
Green sunfish, Lepomis cyanellus	J	W	S	U	6	-	-	-	Tarzwell & Henderson, 1957
Green sunfish, Lepomis cyanellus	J	W	S	U	11	-	-	-	Tarzwell & Henderson, 1957
Green sunfish, Lepomis cyanellus	J	W	S	U	8	8.082	-	-	Tarzwell & Henderson, 1957
Bluegill, Lepomis macrochirus	J	W	S	U	9	-	-	-	Henderson et al., 1959
Bluegill, Lepomis macrochirus	J	W	S	U	17	-	-	-	Macek et al., 1969
Bluegill, Lepomis macrochirus	J	W	S	U	14	-	-	-	Macek et al., 1969
Bluegill, Lepomis macrochirus	J	W	S	U	8.8	-	-	-	Macek et al., 1969
Bluegill, Lepomis macrochirus	J	W	S	U	32	-	-	-	Tarzwell & Henderson, 1957
Bluegill, Lepomis macrochirus	J	W	S	U	18	-	-	-	Tarzwell & Henderson, 1957
Bluegill, Lepomis macrochirus	J	W	S	U	8	-	-	-	Tarzwell & Henderson, 1957
Bluegill, Lepomis macrochirus	J	W	S	U	22	-	-	-	Tarzwell & Henderson, 1957
Bluegill, Lepomis macrochirus	J	W	S	U	3.1	12.16	9.913	9.913	Johnson & Finley, 1980

Appendix A - Dieldrin: (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN- TRATION[d]	LC50/ EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					µg/L	µg/L	µg/L		
Largemouth bass, Micropterus dolomieu	X	W	S	U	3.5	3.5	3.5	3.5	Johnson & Finley, 1980
Channel catfish, Ictalurus punctatus	X	E	S	U	4.5	4.5	4.5	4.5	Johnson & Finley, 1980
SALTWATER SPECIES									
Polychaete worm, Ophryotrocha diadema	L	I	R	U	>100	-	-	-	Hooftman & Vink, 1980
Polychaete worm, Ophryotrocha diadema	A	I	R	U	>100	>100	>100	>100	Hooftman & Vink, 1980
Eastern oyster, Crassostrea virginica	J	E,W	FT	U	34	-	-	-	Butler, 1963
Eastern oyster, Crassostrea virginica	E-L	E,W	S	U	640	-	-	-	Davis & Hidu, 1969
Eastern oyster, Crassostrea virginica	A	E	FT	M	31.20	31.20	31.20	31.20	Parrish et al. 1973
Mysid shrimp, Mysidopsis bahia	A	E	S	U	3.7	-	-	-	U.S. EPA, 1980b
Mysid shrimp, Mysidopsis bahia	A	E	FT	M	4.5	4.5	4.5	4.5	U.S. EPA, 1980b
Sand shrimp, Crangon septemspinosa	A	E	S	U	7	7	7	7	Eisler, 1969
Hermit crab, Pagurus longicarpus	A	E	S	U	18	18	18	18	Eisler, 1969
Grass shrimp, Palaemonetes vulgaris	A	E,W	S	U	50	50	-	-	Eisler, 1969
Grass shrimp, Palaemonetes pugio	A	E,W	FT	M	8.64	8.64	20.78	20.78	Parrish et al. 1973
Korean shrimp,	A	E,W	S	U	16.9	-	-		

Appendix A - Dieldrin: (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		
Korean shrimp, Palaemon macrodactylus	A	E,W	FT	U	6.9	10.80	10.80	10.80	Schoettger, 1970
Pink shrimp, Penaeus duorarum	A	I,E	FT	M	0.70	0.70	0.70	0.70	Parrish et al., 1973
American eel, Anquilla rostrata	J	E	S	U	0.9	0.9	0.9	0.9	Eisler, 1970b
Chinook salmon, Oncorhynchus tshawytscha	J	W	FT	U	1.47	1.47	1.47	1.47	Schoettger, 1970
Atlantic silverside, Menidia menidia	J	W	S	U	5	5	5	5	Eisler, 1970b
Sheepshead minnow, Cyprinodon variegatus	A	E,W	FT	M	10.00	10.00	10.00	10.00	Parrish et al. 1973
Mummichog, Fundulus heteroclitus	A	E,W	S	U	5	-	-	-	Eisler, 1970a
Mummichog, Fundulus heteroclitus	A	E,W	S	U	16	8.944	-	-	Eisler, 1970b
Striped killifish, Fundulus majalis	J	E,W	S	U	5	5	6.687	6.687	Eisler, 1970b
Threespine stickleback, Gasterosteus aculeatus	J	E,W	S	U	15.3	-	-	-	Katz, 1961
Threespine stickleback, Gasterosteus aculeatus	J	E,W	S	U	13.1	14.16	14.16	14.16	Katz, 1961
Striped bass, Morone saxatilis	J	E	FT	U	19.7	19.7	19.7	19.7	Korn & Earnest, 1974
Shiner perch, Cymatogaster aggregata	J	W	S	U	3.7	-	-	-	Earnest & Benville, 1972
Shiner perch, Cymatogaster aggregata	J	W	FT	U	1.50	2.356	2.356	2.356	Earnest & Benville, 1972

Appendix A - Dieldrin: (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		
Dwarf perch, Micrometrus minimus	A	W	S	U	5.0	-	-	-	Earnest & Benville, 1972
Dwarf perch, Micrometrus minimus	A	W	FT	U	2.44	3.493	3.493	3.493	Earnest & Benville, 1972
Bluehead, Thalassoma bifasciatum	A	W	S	U	6	6	6	6	Eisler, 1970b
Striped mullet, Mugil cephalus	A	E	S	U	23	23	23	23	Eisler, 1970b
Northern puffer, Sphaeroides maculatus	A	W	S	U	34	34	34	34	Eisler, 1970b

[a]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 Dieldrin 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, decreased growth of oyster; 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 dieldrin. 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.	SEDIMENT CONCENTRATION, µg/g ORG. CAR.	PORE WATER CONCENTRATION (ug/L)	DOC CONC. (µ/L)	TOC %	Log[a] K_{oc}	REFERENCES
West Bearskin, Minn. *Hyalella azteca*	30.	1.43	48.15	14.9	-	2.97	-	Hoke and Ankley, 1991
	28.	3.75	148.81	42.3	-	2.52	-	
	38.	12.34	474.62	53.9	-	2.60	-	
	45.	30.69	999.67	210.0	-	3.07	-	
	63.	48.73	1450.30	245.5	-	3.36	-	
Pequaywan, Minn. *Hyalella azteca*	13.	14.48	193.58	58.8	-	7.48	-	Hoke and Ankley, 1991
	10.	43.81	541.53	146.6	-	8.09	-	
	-	123.50	1848.80	343.8	-	6.68	-	
	40.	249.30	2280.88	566.1	-	10.93	-	
	55.	479.37	4672.22	518.7	-	10.26	-	
Airport Pond, Minn. (Prelim) *Hyalella azteca*	15.	5.17	304.1	16.70	-	1.70	-	Hoke and Ankley, 1991
	60.	25.24	1493.5	80.12	-	1.69	-	
	100.	97.38	5527.3	89.40	-	1.76	-	
Airport Pond, Minn. *Hyallela azteca*	63	3.77	249.67	13.5	69.69	1.51	4.82	Hoke and Ankley, 1992
	30	16.71	960.34	60.3	73.46	1.74	4.76	
	27	31.55	1889.22	136.0	59.89	1.67	4.78	
	37	61.10	3432.58	224.4	63.73	1.78	4.75	
	53	136.02	7556.11	356.8	67.17	1.80	4.74	
Airport Pond, Minn. (Prelim) *Hyallela azteca*	30	7.29	402.76	30.1	66.0	1.81	4.74	Hoke and Ankley, 1992
	40	30.52	1623.4	143.3	75.5	1.88	4.73	
	47	115.78	6432.2	311.40	65.6	1.80	4.74	
Airport Pond, Minn. *Chironomus tentans*	5	0.09	4.95	-	-	1.82	-	Hoke, 1992
	55	1.00	49.26	-	-	2.03	-	
	50	5.41	252.80	-	-	2.14	-	
	90	12.98	658.88	-	-	1.97	-	
Airport Pond, Minn. *Chironomus tentans*	0	0.05	3.45	-	-	1.45	-	Hoke, 1992
	5	0.10	5.92	-	-	1.69	-	
	50	0.52	36.62	-	-	1.42	-	
	100	3.78	252.00	-	-	1.50	-	
	100	9.64	614.01	-	-	1.57	-	

[a] K_{oc} (L/kg) = sediment concentration (μ/g_{oc}) + calculated free pore water concentration ($\mu g/L$) $\bullet 10^3$ g/kg.