♺EPA

Sediment Quality Criteria for the Protection of Benthic Organisms:

ACENAPHTHENE

CONTENTS

i

FOREWORD

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

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

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

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

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

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

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TABLES

Figure 5-1.   Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to acenaphthene-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 $\div$ ACR)K_{OC}. Symbols for PGMCVs are $\triangle$ for arthropods, $\bigcirc$ for fishes and $\square$ 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 acenaphthene-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 $\div$ ACR)K_{OC}. Symbols for PGMCVs are $\triangle$ for arthropods, $\bigcirc$ for fishes and $\square$ 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 acenaphthene 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 acenaphthene SQC values of 13 μg/g in freshwater sediments having TOC = 10% and 1.3 μg/g in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 23 μg/g when TOC =10% and 2.3 μg/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 acenaphthene in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal line is the saltwater SQC value of 230 μg/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.

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SECTION 1

INTRODUCTION

1.1 GENERAL INFORMATION

Under the Clean Water Act (CWA) the U.S. Environmental Protection Agency (U.S. EPA) is responsible for protecting the chemical, physical and biological integrity of the nation's waters. In keeping with this responsibility, U.S. EPA published ambient water quality criteria (WQC) in 1980 for 64 of the 65 toxic pollutants or pollutant categories designated as toxic in the CWA. Additional water quality documents that update criteria for selected consent decree chemicals and new criteria have been published since 1980. These WQC are numerical concentration limits that are the U.S. EPA's best estimate of concentrations protective of human health and the presence and uses of aquatic life. While these 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 concentration 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 value (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 stable at equilibrium; (2) the concentration in

1-2

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

SQC values may also need to be adjusted because of site specific considerations. In spill situations, where chemical equilibrium between water and sediments has not yet been reached, a sediment chemical concentration less than the 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 total organic carbon (TOC) in the sediment does not greatly affect chemical binding (DeWitt et al., 1992). However, the physical form of the chemical in the sediment may

have an effect. At some sites concentrations in excess of the SQC may not pose risks to benthic organisms, because the compound may be a component of a particulate, such as coal or soot, exceed solubility such as undissolved oil. 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 the Derivation of Site-Specific Sediment Quality Criteria" (U.S. EPA, 1993b). The SQC may be underprotective where the toxicity of other chemicals are additive with the SQC chemical or species of unusual sensitivity occur at the site.

This document presents the theoretical basis and the supporting data relevant to the derivation of the SQC for acenaphthene. 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 By Using Equilibrium Partitioning for the Protection of Benthic Organisms" (U.S. EPA, 1993a) is necessary in order to understand the following text, tables and calculations. Guidance for the acceptable use of SQC values is contained in " Guide for the Use and Application of Sediment Quality Criteria for Nonionic Organic Chemicals" (U.S. EPA, 1993c).

1.2 GENERAL INFORMATION: ACENAPHTHENE:

Acenaphthene is a member of the polycyclic aromatic hydrocarbon (PAH) group of organic compounds. It occurs both naturally in coal tar, and as a by product of manufacturing processes such as petroleum refining, shale oil processing and coal tar distilling (Verschueren, 1983). Other man made sources of acenaphthene include its generation as a by product of the

combustion of tobacco, and its presence in asphalt and in soots generated by the combustion of aromatic fuels amended with pyridine (Verschueren, 1983). Acenaphthene is used in manufacturing processes to produce dyes, plastics, insecticides and fungicides (Windholtz et al., 1983). Some PAHs are of environmental concern because they are known to be carcinogens and/or mutagens (Brookes, 1977). With an increase in fossil fuel consumption in the United States an increase in emissions of PAHS to the environment can be expected over the next several decades (Eadie et al., 1982).

Acenaphthene has a two ring bridged structure (Figure 1-1). It has a solubility in water at 25°C of 3.94 mg/l (Miller et al., 1985), and is a solid at room temperature (melting point of 116°C). Two significant processes which can influence the fate of acenaphthene in sediment are sorption and biodegradation (U.S. EPA, 1980). Sorption of acenaphthene onto solids in the water column and subsequent settling, as well as partitioning onto organics in the sediment, can significantly affect acenaphthene transport. Bioaccumulation is a short-term process in which PAHs with 4 rings or less are metabolized and long-term partitioning into biota is not considered a significant fate process (U.S. EPA, 1980). Other processes found to have little or no effect on the fate of acenaphthene in the sediment are oxidation, hydrolysis and volatilization (U.S. EPA, 1980).

The acute toxicity of acenaphthene from individual toxicity tests ranges from 120.0 to 2,045 μg/L for freshwater and 160 to 16,440 μg/L for saltwater organisms (Appendix A). Differences between concentrations of acenaphthene causing acute lethality and chronic toxicity are small; acute-chronic ratios range from 1.5 to 6.7 (Table 3-3). Although acenaphthene bioaccumulates in aquatic biota, the associated health or ecological risks are unknown.
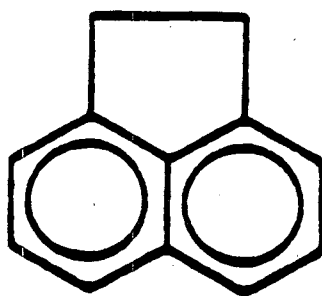
MOLECULAR FORMULA	$C_{12}H_{10}$
MOLECULAR WEIGHT	154.21
DENSITY	1.069 g/cc @ 20°C
MELTING POINT	90-95°C
PHYSICAL FORM	Orthorhombic bipyramidal needles
VAPOR PRESSURE	0.0026 mPa (25°C)

| CAS NUMBER: | 83-32-9 |
| CHEMICAL NAME: | 1,2-Dihydroacenaphthylene; periethylenenaphthalene; 1,8-ethylenenaphthalene |

Figure 1-1. Chemical structure and physical-chemical properties of acenaphthene.

1.3 OVERVIEW OF DOCUMENT:

Section 1 provides a brief review of EqP methodology, and a summary of the physical-chemical properties and aquatic toxicity of acenaphthene. Section 2 reviews a variety of methods and data useful in deriving partition coefficients for acenaphthene and includes the K_{oc} recommended for use in the derivation of the acenaphthene SQC. Section 3 reviews aquatic toxicity data contained in the acenaphthene WQC document (U.S. EPA, 1980) and new data that were used to derive the Final Chronic Value (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 acenaphthene in the derivation of the SQC. Section 4 reviews data on the toxicity of acenaphthene in sediments, the need for organic carbon normalization of acenaphthene 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 acenaphthene and its uncertainty. The SQC for acenaphthene is then compared to STORET (U.S. EPA, 1989b) and National Status and Trends (NOAA, 1991) data on acenaphthene's environmental occurrence in sediments. Section 6 concludes with the criteria statement for acenaphthene. 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 non-ionic 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 methodology for deriving SQC. The methodology is discussed in detail in the "Technical Basis for Deriving Numerical National Sediment Quality Criteria for Nonionic Organic Contaminants by Using Equilibrium Partitioning for the Protection of Benthic Organisms" (U.S. EPA, 1993a), hereafter referred to as the SQC Technical Basis Document.

Bioavailability of a chemical at a particular sediment concentration often differs from one sediment type to another. Therefore, a method is necessary for determining a SQC based on the bioavailable chemical fraction in a sediment. For nonionic organic chemicals, the concentration-response relationship for the biological effect of concern can most often be correlated with the interstitial water (i.e., pore water) concentration (μg chemical/liter pore

water) and not to the sediment chemical concentration (μg chemical/g sediment) (Di Toro et al., 1991). From a purely practical point of view, this correlation suggests that if it were possible to measure the pore water chemical concentration, or predict it from the total sediment concentration and the relevant sediment properties, then that concentration could be used to quantify the exposure concentration for an organism. Thus, knowledge of the partitioning of chemicals between the solid and liquid phases in a sediment is a necessary component for establishing SQC. It is for this reason that the methodology described below is called the equilibrium partitioning (EqP) method.

It is shown in the SQC Technical Basis Document (U.S. EPA, 1993a) that the final acute values (FAVs) in the WQC documents are appropriate for benthic species for a wide range of chemicals. (The data showing this for acenaphthene 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:

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

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 acenaphthene 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 and the weight fraction of organic carbon in the sediment (f_{oc}). The relationship is as follows:

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

It follows that:

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

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 acenaphthene.

2.2 DETERMINATION OF K_{ow} FOR ACENAPHTHENE:

Several approaches have been used to determine K_{ow} for 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}'s. The measurement methods were shake-centrifugation (SC) and 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.

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

METHOD	$LOG_{10}K_{ow}$	REFERENCE
Measured	3.92	Banerjee et al., 1980
Estimated	3.70	Yalkowsky et al., 1983
Estimated	3.88	SPARC[a]
Estimated	3.92	Miller et al., 1985
Estimated	3.98	Mabey et al., 1982
Estimated	4.03	Yalkowsky et al., 1979
Estimated	4.07	CLOGP[b]
Estimated	4.15	Mackay et al., 1980
Estimated	4.22	Kamlet et al., 1988
Estimated	4.33	Callahan et al., 1979
Estimated	4.43	Arbuckle 1983

[a]SPARC is from SPARC Performs Automated Reasoning in Chemistry, (Karickhoff et al., 1989).
[b]CLOGP is an algorithm that is included in the database QSAR located at the U.S. EPA, Environmental Research Lab., Duluth, MN (Chou and Jurs, 1979).

In an examination of the literature data for acenaphthene, only one primary reference was found, with a measured $log_{10}K_{ow}$ value of 3.92 (Banerjee et al., 1980) (Table 2-1). Several primary references were found for estimated $log_{10}K_{ow}$ values in the literature ranging from 3.70 to 4.43 (Table 2-1). Although the range of reported values for acenaphthene is significantly

lower than the range of values for some other compounds, it is relatively large, and we were not able to determine from studying the primary articles that any value was more likely to be accurate than any other.

K_{ow} values for SPARC and CLOGP are also included in Table 2-1. SPARC is a computer expert system under development at ERL,A, and the University of Georgia, at Athens. The CLOGP algorithm is included in the database QSAR located at EPA's Environmental Research Laboratory (ERL,D) at Duluth, Minnesota. For more information on SPARC and CLOGP see U.S. EPA (1993a). The SPARC estimated $\log_{10}K_{ow}$ value for acenaphthene is 3.88. The CLOGP program estimate of the $\log_{10}K_{ow}$ value for acenaphthene using structure activity relationships is 4.07.

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} acenaphthene (Table 2-2). The SC method yielded a $\log_{10}K_{ow} = 3.84$ (n=4), the GCol method yielded a $\log_{10}K_{ow} = 4.17$ (n=4), and the SSF method yielded a $\log_{10}K_{ow} = 3.83$ (n=3). 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 3.83 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 three K_{ow} measurements made by SSF. The logs of the K_{ow} values measured by SSF range from 3.81 to 3.84.

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

SHAKE-CENTRIFUGATION	GENERATOR COLUMN	SLOW STIR FLASK
3.82	4.18	3.81
3.84	4.17	3.84
3.88	4.16	3.84
3.84	4.17	
3.84	4.17	3.83

*Log_{10} of mean measured values.

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 acenaphthene, sediment organic carbon (OC) and non-dissolved organic carbon (DOC) associated acenaphthene 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).

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

$$K_p = \frac{f_{oc}\, K_{oc}}{1 + mf_{oc}\, K_{oc}\, /\; v_x} \tag{2-4}$$

where m is the particle concentration in the suspension (kg/L), and $v_x = 1.4$, an empirical constant.

In this expression the K_{oc} is given by:

$$\log_{10}K_{oc} = 0.00028 + 0.983 \log_{10}K_{ow} \tag{2-5}$$

A sorption isotherm experiment that demonstrates the effect of particle suspensions was found in a comprehensive literature search for partitioning information for acenaphthene (Table 2-2) (Mihelcic and Luthy, 1988). The experiment with four different concentrations of particles in suspension showed an observed Kp of 52 L/kg for an acenaphthene solution and soil (2.9 $\pm$ 0.28% organic carbon). Calculated Kp using K_{oc} (Equation 2-5) and f_{oc} is 175 L/kg. The difference between the observed and calculated Kp can be explained by particle interaction effects. Particle interaction results in a lower observed partition coefficient. The particle interaction model (Equation 2-4) predicts Kp of 36.8 L/kg to 6.7 L/kg for respective solids

concentrations of 0.03 kg/L to 0.20 kg/L which is in order with the observed Kp. $Log_{10}K_{oc}$ computed from observed K_p and f_{oc} is 3.25. This value is lower than K_{oc} from laboratory measurements as a result of particle effects. This data is presented as an example of particle effects only, as 100 percent reversibility is assumed in the absence of a desorption study and an actual K_{oc} can not be computed.

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

Observed $Log_{10}K_{oc}$	n	Solids (kg/L)	Reference
3.25	1	0.03-0.20	Milhelcic and Luthy, 1988

In the absence of particle effects, K_{oc} is related to K_{ow} via Equation 2-5, shown above. For $log_{10}K_{ow} = 3.83$ (ERL,A mean measured value), this expression results in an estimate of $log_{10}K_{oc} = 3.76$

2.3.2 K_{oc} FROM SEDIMENT TOXICITY TESTS:

Measurements of K_{oc} are available from sediment toxicity tests using acenaphthene (Swartz, 1991). These tests are from three marine saltwater sediments having a range of organic carbon contents of 1.02 to 4.37 percent (Table 4-1; Appendix B). Acenaphthene concentrations were measured in the sediments and pore waters providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. The pore water measurements did not

increase at values greater than 1,000 μg/L suggesting that the limit of aqueous solubility was being approached. These tests were run at 15°C, but a literature search for acenaphthene revealed no solubility data at this temperature. Solubility for acenaphthene is reported as 3.94 mg/L at 25°C (Miller et al., 1985), supporting the idea of saturation limitation. As a result, computations for the partition coefficient did not include treatments where pore water concentrations were greater than 1,000 μg/L.

The upper panel of Figure 2-1 is a plot of the organic carbon-normalized sorption isotherm for acenaphthene where the sediment acenaphthene concentration (μg/g_{oc}) is plotted versus pore water concentration (μg/L). The data used to make this plot are included in Appendix B. Data from treatments where pore water concentrations were greater than 1,000 μg/L were not included on the plot. The line of unity slope corresponding to the $\log_{10}K_{oc} = 3.76$ is compared to the data. A probability plot of the observed experimental $\log_{10}K_{oc}$ values is shown in lower panel of Figure 2-1. The $\log_{10}K_{oc}$ values are approximately normally distributed with a mean of $\log_{10}K_{oc} = 3.58$ and a standard error of the mean of 0.012. This value is statistically indistinguishable from $\log_{10}K_{oc} = 3.76$, which was computed from the experimentally determined acenaphthene $\log_{10}K_{ow}$ of 3.83 Equation 2-5.

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

The K_{oc} selected to calculate the sediment quality criteria for acenaphthene is based on the regression of $\log_{10}K_{oc}$ from $\log_{10}K_{ow}$ (Equation 2-5), using the acenaphthene $\log_{10}K_{ow}$ of 3.83 recently measured by ERL,A. This approach rather than the use of the K_{oc} from the toxicity test was adopted because the regression equation is based on the most robust data set
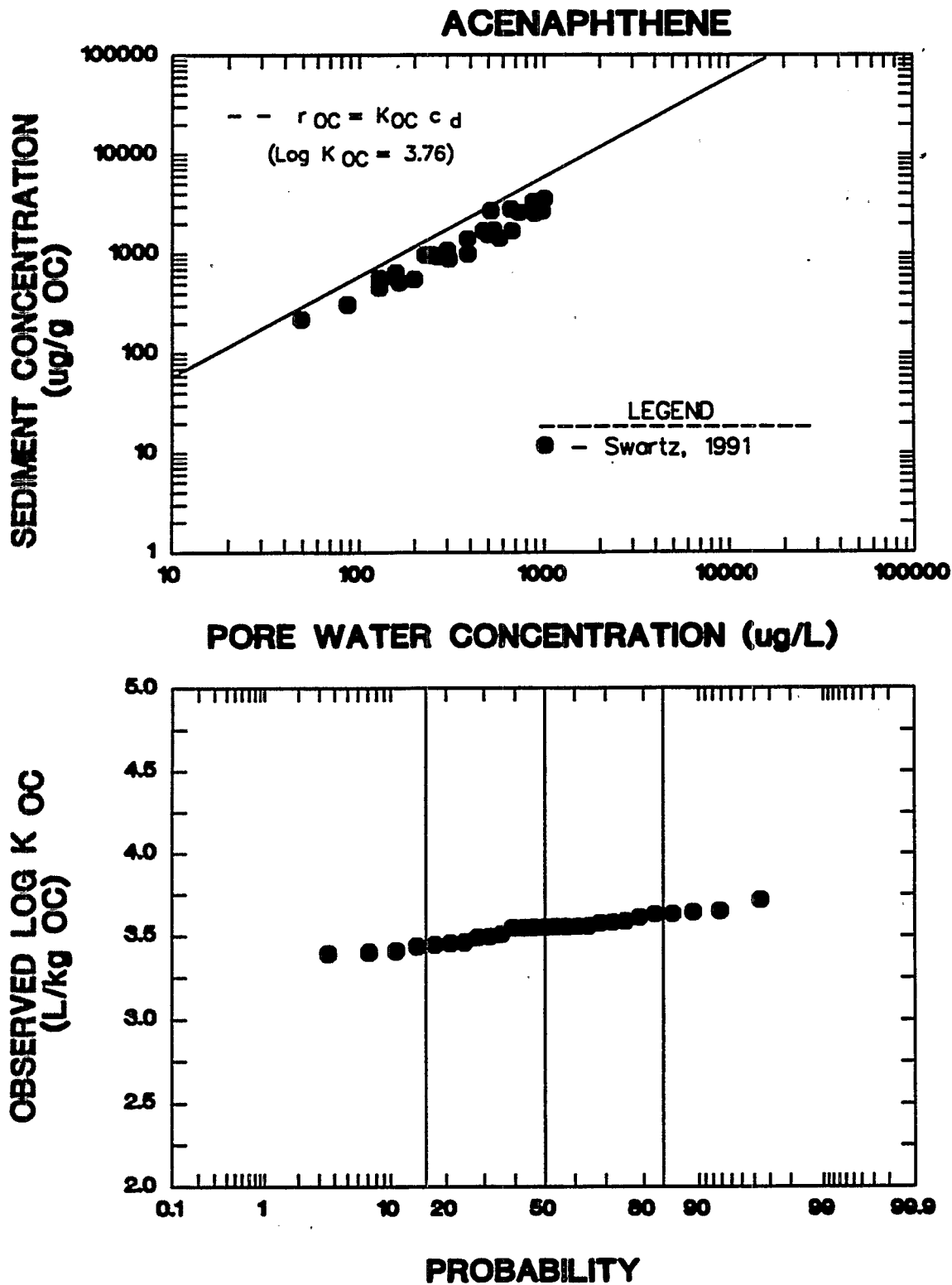
ACENAPHTHENE

Figure 2-1. Organic carbon-normalized sorption isotherm for acenaphthene (top) and probability plot of Koc (bottom) from sediment toxicity tests conducted by Swartz (1991). The line in the top panel represents the relationship predicted with a log K_{oc} of 3.76, that is $C_{s,oc} = K_{oc} \cdot C_d$.

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 $\log_{10}K_{oc} = 3.76$. This value is in very good agreement with the $\log_{10}K_{oc}$ of 3.58 measured in the sediment toxicity tests.

SECTION 3

TOXICITY OF ACENAPHTHENE: WATER EXPOSURES

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

The equilibrium partitioning (EqP) method for derivation of sediment quality criteria (SQC) uses the acenaphthene water quality criteria (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 acenaphthene WQC document (U.S. EPA, 1980) is revised using new aquatic toxicity test data, and (2) the use of this FCV is justified as the effects concentration for SQC derivation.

3.2 ACUTE TOXICITY-WATER EXPOSURES:

Twenty standard acute toxicity tests with acenaphthene have been conducted on 10 freshwater species from 10 genera (Appendix A). Overall genus mean acute values (GMAVs) range from 120 to 2,045 μg/L. Three invertebrates and two fishes were among the most sensitive species; overall GMAVs for these taxa range from 120 to 670 μg/L. Tests on the benthic life-stages of 5 species from 5 genera are contained in this database (Figure 3-1; Appendix A). Benthic organisms were among both the most sensitive, and most resistant, freshwater genera to acenaphthene; GMAVs range from 240 and >2,040 μg/L. Three

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, N = nymph, X = unspecified life stage.

epibenthic species, stoneflies, a snail and channel catfish, were tested. Two infaunal species were tested, including the amphipod, Gammarus minus (LC50 = 460 μg/L), and the midge, Paratanytarsus sp. (LC50 = 2,045 μg/L). The FCV derived from the overall GMAVs (Stephan et al., 1985) for freshwater organisms is 80.01 (Table 3-2).

Twenty-one acute toxicity tests have been conducted on 10 saltwater species from 10 genera (Appendix A). Overall GMAVs range from 245.0 to 8,163 μg/L., similar to the range for freshwater genera. Crustaceans were most sensitive; GMAVs range from 245.0 to 1,125 μg/L. Benthic life-stages from 6 species from 6 genera have been tested (Figure 3-2; Appendix A). They are among both the most sensitive, and most resistant, saltwater genera to acenaphthene. The most sensitive benthic species is the sand shrimp, Crangon septemspinosus, with a 96-hour LC50 of 245.0 μg/L based on unmeasured concentrations. The mysid, Mysidopsis bahia, has a similar sensitivity with an average, flow-through 96-hour LC50 of 317.7 μg/L based on measured concentrations. Other benthic species for which there are data appear less sensitive; GMAVs range from 589.4 to 7,693 μg/L. The FAV derived from the overall GMAVs (Stephan et al., 1985) for saltwater organisms is 140.8 μg/L (Table 3-2).

3.3 CHRONIC TOXICITY - WATER EXPOSURES:

Chronic life-cycle toxicity tests have been conducted with the freshwater midge (Paratanytarsus sp.) and the saltwater mysid (M. bahia) (Table 3-1) and early life stage tests have been conducted with the fathead minnow (Pimephales promelas) and sheepshead minnow (Cyprinodon variegatus) (Table 3-1). For each of these species, except for P. promelas, one or more benthic life stages were exposed to acenaphthene. Other chronic toxicity tests have been
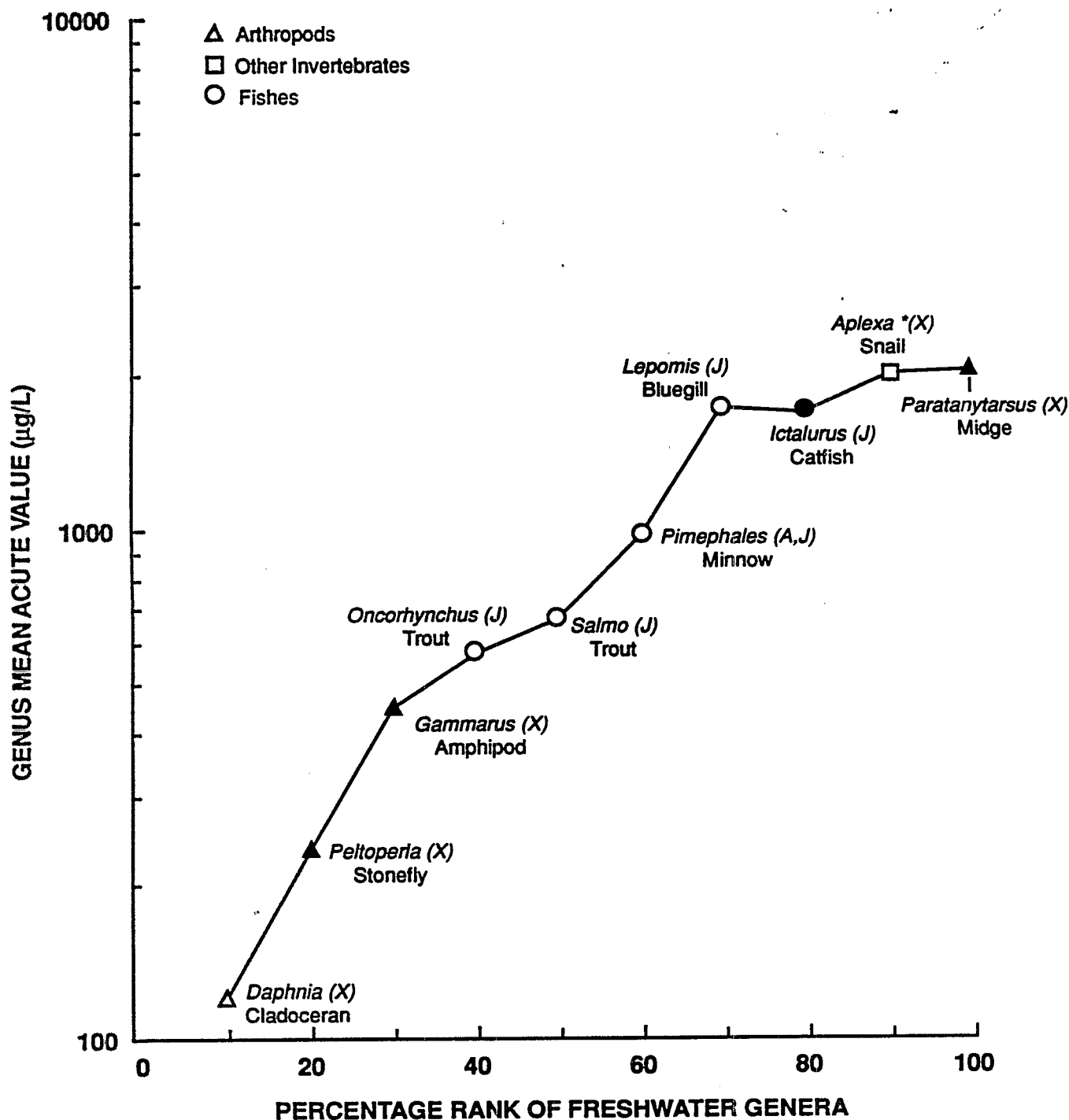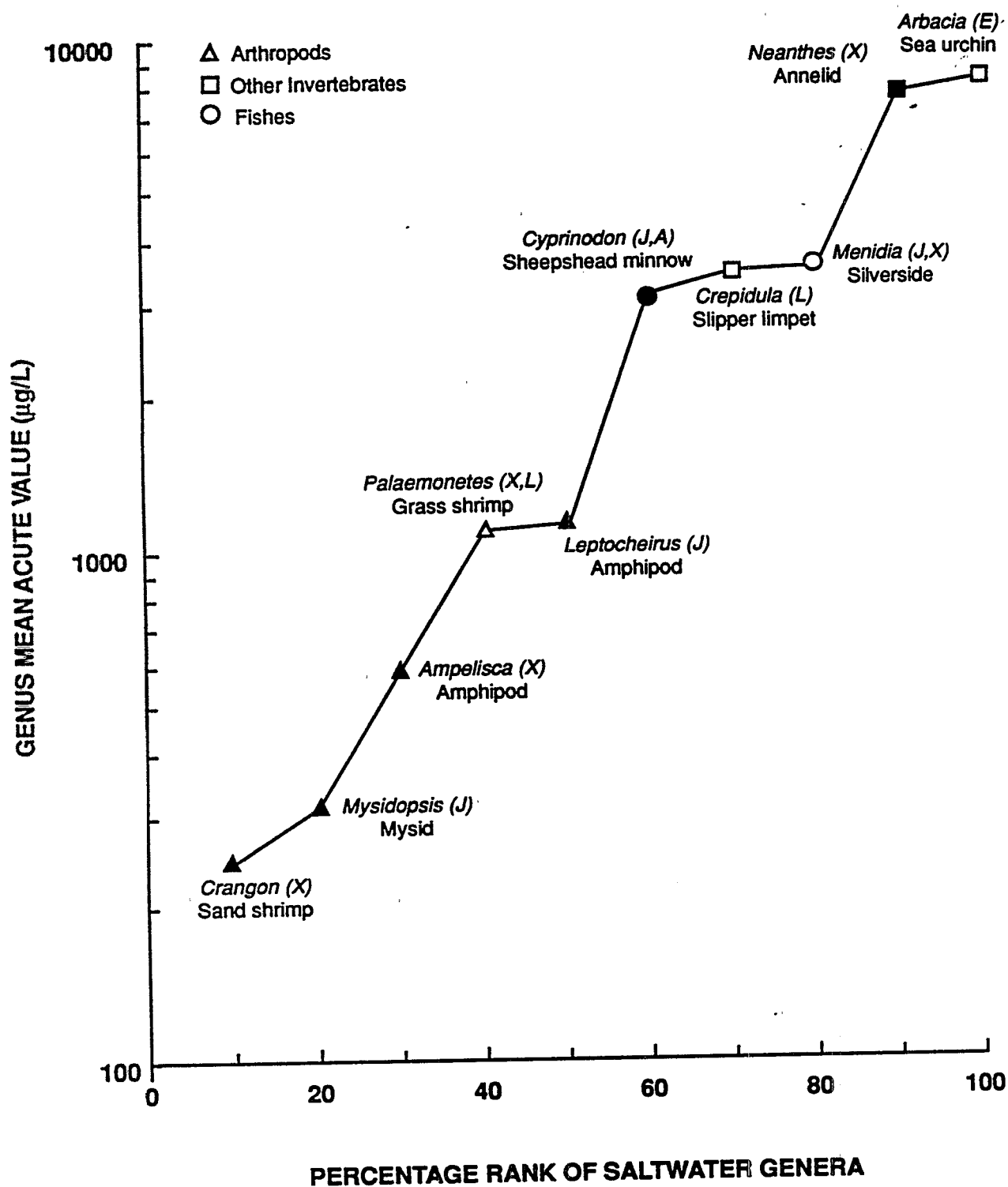
Figure 3-2. Genus mean acute values from water only acute toxicity tests using saltwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. E = embryo, J = juvenile.

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

Common Name, Scientific Name	Test[a]	Habitat[b] (Life stage)	NOEC[c]	Parental LOEC[c]	Parental Effect[d]	Progeny LOEC[c]	Progeny Effect[d]	References
			µg/L	µg/L		µg/L		
FRESHWATER SPECIES								
Midge, Paratanytarsus sp.	LC	E	32 - 295	575	59% G 85% M	575	100% M	Northwestern Aquatic Sciences, 1982
Midge, Paratanytarsus sp.	LC	E	27 - 164	315 676	21% M 62% M	676	64% M	Northwestern Aquatic Sciences, 1982; Thursby, 1991a
Fathead minnow, Pimephales promelas	ELS	W	50	109 410 630	5% G 26% G;45% M 100% M	-	-	Academy of Natural Sciences, 1981; Thursby, 1991a
Fathead minnow, Pimephales promelas	ELS	W	50, 109	410 630	20% G;66% M 100% M	-	-	Academy of Natural Sciences, 1981
Fathead minnow, Pimephales promelas	ELS	W	67-332	495	54% G	-	-	Cairns and Nebeker, 1982
Fathead minnow, Pimephales promelas	ELS	W	197, 345	509 682 1,153	30% G 52% G;45% M 87% G;97% M	-	-	Cairns and Nebeker, 1982
Fathead minnow, Pimephales promelas	ELS	W	64	98 149 271 441	24% M 65% M 75% M 80% M	-	-	ERCO, 1981
Fathead minnow, Pimephales promelas	ELS	W	50, 91	139 290 426	20% M 50% M 52% M	-	-	ERCO, 1981

TABLE 3-1. Acenaphthene (Cont'd)

Common Name, Scientific Name	Test[a]	Habitat[b] (Life stage)	NOEC[c]	Parental LOEC[c]	Parental Effect[d]	Progeny LOEC[c]	Progeny Effect[d]	References
			µg/L	µg/L		µg/L		
SALTWATER SPECIES								
Mysid, Mysidopsis bahia	LC	B (J,A)	100-240	340 510	93% R 100% M	>340	-	Horne et al., 1983
Mysid, Mysidopsis bahia	LC	B (J,A)	0.394*-44.6	91.8 168 354	91% R 100% R;34% G 100% R;96% M	-	-	Thursby et al., 1989b
Sheepshead minnow, Cyprinodon variegatus	ELS	B (E) B, W (J,A)	240-520	970 2,000 2,800	70% M 100% M 100% M	-	-	Ward et al., 1981

[a]TEST: LC = lifecycle, PLC = partial lifecycle, ELS = early lifestage

[b]HABITAT: I = infauna, B = 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.

*Carrier control

3-6

conducted with these two freshwater species (Lemke, 1984; Lemke et al.,1983; Lemke and Anderson, 1984) but insufficient documentation is available to permit use of these results (Thursby, 1991a).

Two acceptable life cycle toxicity tests have been conducted with Paratanytarsus sp. (Northwestern Aquatic Sciences, 1981). In the first test there was a 59% reduction in growth and an 85% reduction in reproduction in 575 μg/L relative to control animals (Table 3-2). Eggs produced by animals in this first test failed to hatch at 575 μg/L. There was no significant effect on parents or egg hatchability in acenaphthene concentrations from 32 to 295 μg/L. In the second test with Paratanytarsus sp. there was a 21% reduction in survival in 315 μg/L relative to control animals; egg hatchability was not affected at the highest concentration tested (676 μg/L); although survival of hatched Paratanytarsus sp. larvae was reduced 64% in this concentration.

A total of six early life-stage toxicity tests have been conducted with the P. promelas part of a round-robin test series; two each from three laboratories (Table 3-2). The effect concentrations across laboratories and tests ranged from 98 to 509 μg/L, a factor of 5.2. Growth (dry weight), survival or both growth and survival were reduced. Only one of these test pairs had a suitable measured acute value, allowing calculation of an acute-chronic ratio (Cairns and Nebeker, 1982). The concentration-response relationships were similar for these two tests. Parental fish were unaffected in the first test at acenaphthene concentrations ranging from 67 to 332 μg/L, while fish exposed to 495 μg/L had a 54% reduction in growth relative to control fish. In the second test, Cairns and Nebeker (1982) observed a 30% reduction in growth in parental fish in 509 μg/L while there was no effect on fish exposed to 197 to 345 μg/L.

TABLE 3-2. - SUMMARY OF FRESHWATER AND SALTWATER ACUTE AND CHRONIC VALUES, ACUTE-CHRONIC RATIOS AND DERIVATION OF FINAL ACUTE VALUES, FINAL ACUTE-CHRONIC RATIOS AND FINAL CHRONIC VALUES FOR ACENAPHTHENE.

Common Name, Scientific Name	Acute Value (µg/L)	Chronic Value (µg/L)	Acute-Chronic Ratio	Species Mean Acute-Chronic Ratio
FRESHWATER SPECIES				
Fathead minnow, Pimephales promelas	608	405.4	1.500	--
Fathead minnow, Pimephales promelas	608	419.1	1.451	1.475
Fathead minnow, Pimephales promelas	--	73.82	--	--
Fathead minnow, Pimephales promelas	--	211.4	--	--
Fathead minnow, Pimephales promelas	--	79.20	--	--
Fathead minnow, Pimephales promelas	--	112.5	--	--
Midge, Paratanytarsus sp.	2,045[a]	411.9	4.965	--
Midge, Paratanytarsus sp.	2,045[a]	227.3	8.997	6.683
SALTWATER SPECIES				
Mysid, Mysidopsis bahia	466.1	285.7	1.631	--
Mysid, Mysidopsis bahia	460	63.99	7.189	3.424
Sheepshead minnow, Cyprinodon variegatus	3,100	710.2	4.365	4.365

TABLE 3-2. Acenaphthene (Cont'd)

[A]Geometric mean of two flow-through measured tests from the same laboratory.

Freshwater:

Final Acute Value = 80.01 µg/L
Final Acute-chronic Ratio = 3.484
Final Chronic Value = 22.96 µg/L

Saltwater:

Final Acute Value = 140.8 µg/L
Final Acute-chronic Ratio = 3.484
Final Chronic Value = 40.41 µg/L

Data from three saltwater chronic toxicity tests are available, two with M. bahia and one with C. variegatus. M. bahia reproduction was affected by acenaphthene in two tests from two different laboratories. In the first test (Horne et al., 1983) there was a 93% decrease in reproduction in 340 μg/L relative to control M. bahia; all M. bahia in 510 μg/L died. No effects were observed at 100 or 240 μg/L in the parental generation, and juveniles released in $\geq$ 340 μg/L were not affected. In the second test (Thursby et al., 1989b) there was a 34% decrease in growth in 168 μg/L and 96% increase in mortality at 354 μg/L. There was a 91% decrease in reproduction in M. bahia exposed to 91.8 μg/L and M. bahia exposed to 168 and 354 μg/L did not reproduce. M. bahia exposed to $\leq$ 44.6 μg/L were not affected.

C. variegatus exposed to acenaphthene in an early life stage test (Ward et al., 1981) were affected at acenaphthene concentrations of $\geq$970 μg/L (Table 3-2). There was a 70% reduction in survival of fish hatched in 970 μg/L. Fewer than 10% of the embryos at $\geq$ 2,000 μg/L hatched and all fish that hatched died. There was no effect on either survival or growth in fish exposed to 240 or 520 μg/L.

The difference between acute and chronic toxicity of acenaphthene is small (Table 3-2). Species mean acute-chronic ratios are 1.475 for P. promelas, 3.424 for M. bahia, 4.365 for C. variegatus and 6.683 for Paratanytarsus sp. The final ACR, the geometric mean of these four values, is 3.484.

The FCV (Table 3-2) are used as the effect concentrations for calculating the SQC for protection of benthic species. The FCV for freshwater organisms of 22.96 μg/L is the quotient of the FAV of 80.01 μg/L and the final ACR of 3.484. Similarly, the FCV for saltwater organisms of 40.41 μg/L is the quotient of the FAV of 140.8 μg/L and the final ACR of 3.484.

3.4 APPLICABILITY OF THE WATER QUALITY CRITERION AS THE EFFECTS CONCENTRATION FOR DERIVATION OF THE ACENAPHTHENE 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 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 acenaphthene-specific comparisons in sensitivity of benthic and all (benthic and water-column) species conducted. The following paragraphs examine the data on the similarity of sensitivity of benthic and all species for acenaphthene.

For acenaphthene, benthic species account for 4 out of 10 genera tested in freshwater, and 6 out of 10 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 acenaphthene was performed using the Approximate Randomization method (Noreen, 1989).

3-11

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 acenaphthene, the test-statistic falls at the 33 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

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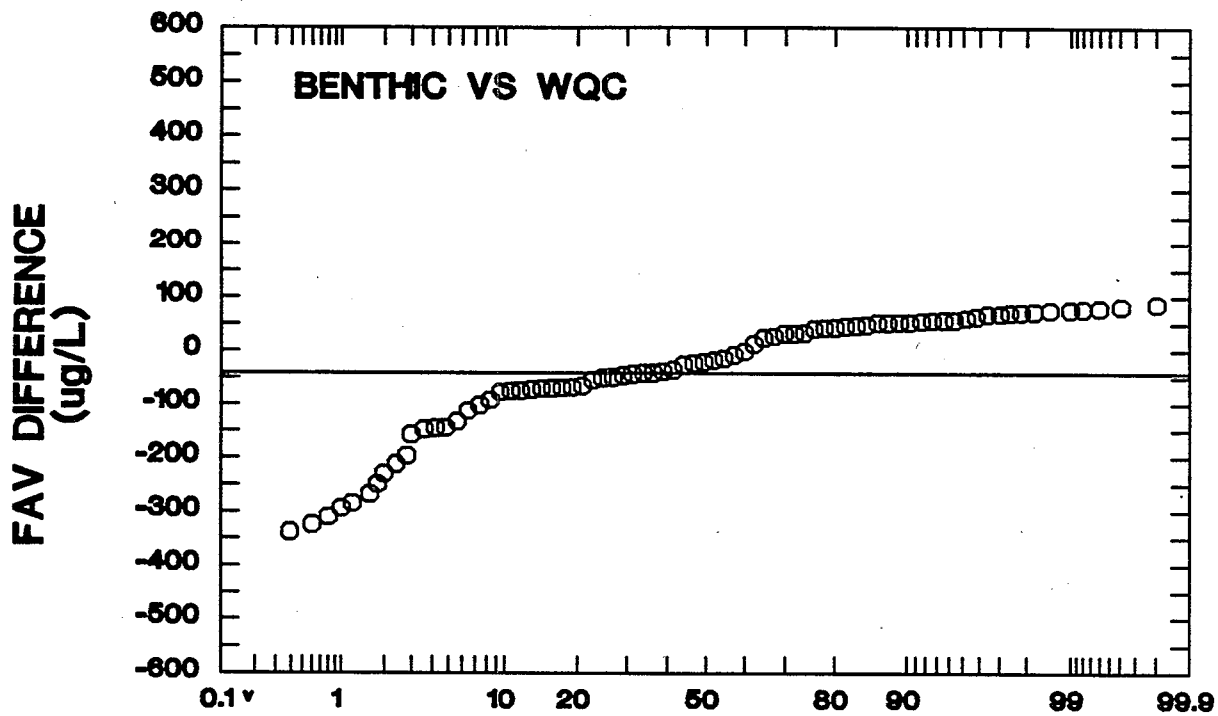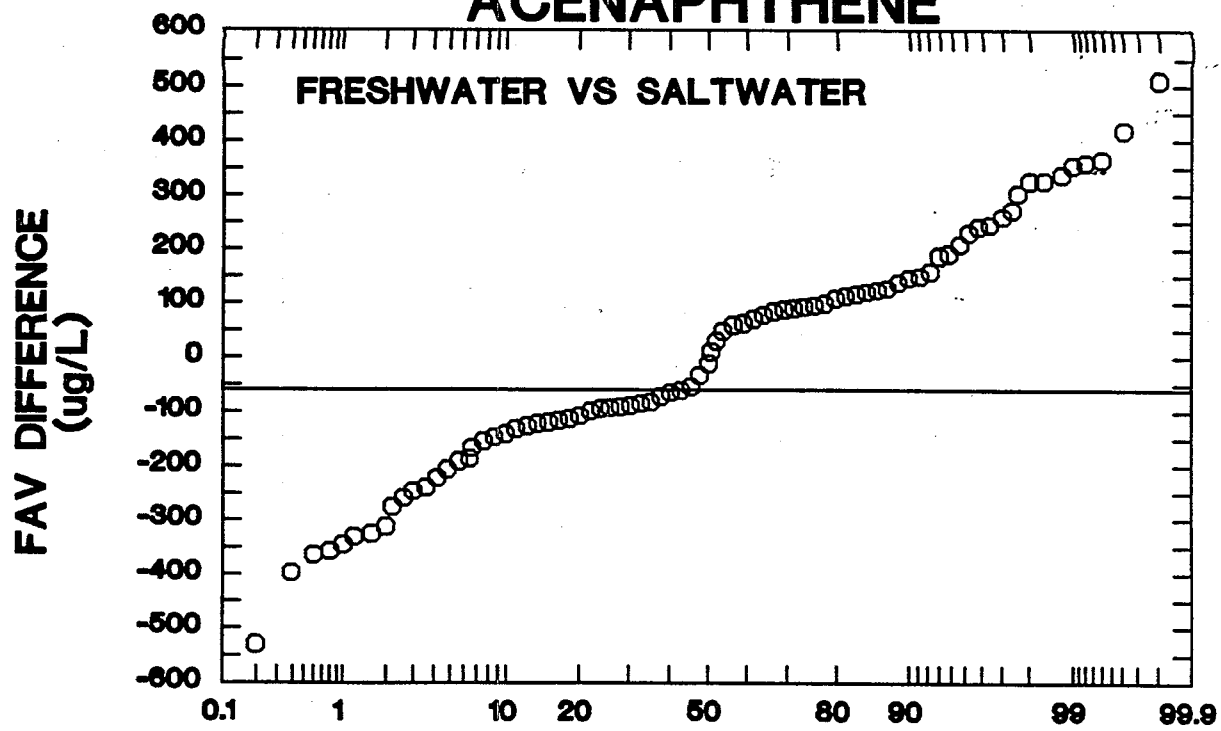
Figure 3-3. Probability distribution of FAV difference statistics to compare water-only data from freshwater vs. saltwater (upper panel) and benthic vs. WQC (lower panel) data.

3-13

TABLE 3-3. RESULTS OF APPROXIMATE RANDOMIZATION TEST FOR
THE EQUALITY OF THE FRESHWATER AND SALTWATER-LC50
DISTRIBUTIONS FOR ACENAPHTHENE AND APPROXIMATE
RANDOMIZATION TEST FOR THE EQUALITY OF BENTHIC AND
COMBINED BENTHIC AND WATER COLUMN (WQC) LC50
DISTRIBUTIONS.

Comparison	Habitat or Water Type[a]		AR Statistic[b]	Probability[c]
Fresh vs Salt	Fresh (10)	Salt (10)	-59.04	33
Benthic vs Water Column + Benthic (WQC)	Benthic (10)	WQC (20)	-41.34	31

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

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 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

3-14

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 31 percentile and the hypothesis of no difference in sensitivity is accepted (Table 3-3). This analysis suggests that the FCV for acenaphthene based on data from all tested species is an appropriate effects concentration for benthic organisms.

SECTION 4

TOXICITY OF ACENAPHTHENE (ACTUAL AND PREDICTED): SEDIMENT EXPOSURES

4.1 TOXICITY OF ACENAPHTHENE IN SEDIMENTS:

The toxicity of acenaphthene spiked into sediments has been tested with two saltwater amphipod species. Freshwater benthic species have not been tested in acenaphthene-spiked sediments. All concentrations of acenaphthene in sediments or interstitial water where effects were observed in benthic species (Table 4-1) are greater than SQC or FCV concentrations reported in this document. Details about exposure methodology are provided because, unlike aquatic toxicity tests, sediment testing methodologies have not been standardized. Generalizations across species or sediments are limited because of the limited number of experiments. Therefore, insights into relative sensitivities of aquatic species to acenaphthene can only be obtained from results of water-only tests (Section 3). Data are available from many experiments using both field and laboratory sediments contaminated with mixtures of PAHs and other compounds which include acenaphthene. Data from these studies have not been included here because it is not possible to determine the contribution of acenaphthene to the observed toxicity.

Swartz (1991) exposed the amphipods Eohaustorius estuarius and Leptocheirus plumulosus to three acenaphthene-spiked sediments with total organic carbon content (TOC) ranging from 0.82 to 4.21%. Sediments were rolled (1) for four hours in acenaphthene-coated

4-1

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

Common/Sci. Name	Sediment Source, Description	TOC (%)	Method[b]/Duration (Days)	Response	Sediment Acenaphthene LC50, µg/g		Pore Water LC50, µg/L	References
					Dry wt.	Org. Car.		
SALTWATER SPECIES								
Amphipod, Eohaustorius estuarius	South Beach, OR	1.23 0.82[a]	S,M/10	LC50	44.4	4,330	800	Swartz, 1991
Amphipod, Eohaustorius estuarius	McKinney Slough, OR	2.49	S,M/10	LC50	47.8	1,920	609	Swartz, 1991
Amphipod, Eohaustorius estuarius	Eckman Slough, OR	4.21	S,M/10	LC50	68.4	1,630	542	Swartz, 1991
Amphipod, Leptocheirus plumulosus	South Beach, OR	1.62 0.82[a]	S,M/10	LC50	>193	>23,500	>1,720	Swartz, 1991
Amphipod, Leptocheirus plumulosus	McKinney Slough, OR	2.52	S,M/10	LC50	193	7,730	1,410	Swartz, 1991
Amphipod, Leptocheirus plumulosus	Eckman Slough, OR	3.66 2.97[a]	S,M/10	LC50	382	11,200	1,490	Swartz, 1991

[a] TOC of sediment used in highest concentration(s).

[b] S = Static; M = Measured.

bottles, (2) stored at 4°C for either 72 hours (experiments with E. estuarius) or overnight (experiments with L. plumulosus), (3) rolled for an additional four hours, and (4) then stored for 7 days at 4°C. The 10-day LC50's for both species increased slightly with increasing organic carbon concentration when the acenaphthene concentration was expressed on a dry weight basis, but decreased for E. estuarius and no pattern was apparent for L. plumulosus when concentration was expressed on an organic carbon basis. LC50's normalized to dry weight differed by less than a factor of 1.5 (44.4 to 68.4 μg/g) for E. estuarius and less than a factor of 1.9 for L. plumulosus over a 5.3-fold range of TOC. The organic carbon normalized LC50's for E. estuarius differed by a factor of 2.7 (1,630 to 4,330 μg/g_{oc}) while for L. plumulosus they differed by a factor of > 3.0 (7730 to > 23,500 μg/g_{oc}).

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

4.2 CORRELATION BETWEEN ORGANISM RESPONSE AND PORE WATER CONCENTRATION:

One corollary of the EqP theory is that pore-water LC50's for a given organism should

be constant across sediments of varying organic carbon content (U.S. EPA, 1993a). Appropriate pore-water LC50 values are available for two benthic species (Table 4-2). Swartz (1991) found 10-day LC50 values based on pore-water concentrations varied by a factor of 1.5 (542 to 800 μg/L) for E. estuarius and by a factor of > 1.2 (1,410 to > 1,720 μg/L) for L. plumulosus. This variability is somewhat less than that shown when either dry weight (factors of 1.5 and 1.9) or organic carbon (factors of 2.7 and 3.0) normalization are used to determine LC50s based on acenaphthene concentration in sediments.

A more detailed evaluation of the degree to which the response of benthic organisms can be predicted from toxic units of substances in pore water can be made utilizing results from toxicity tests with sediments spiked with other substances, including acenaphthene and phenanthrene (Swartz, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989), dieldrin (Hoke, 1992), fluoranthene (Swartz et al., 1990, De Witt 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 utilize designs at least as rigorous as those conducted at the EPA laboratories. Tests with acenaphthene and phenanthrene used two saltwater amphipods (L. plumulosus and E. estuarius) and marine sediments. Tests with fluoranthene used the saltwater amphipod (Rhepoxynius abronius) and marine sediments. Freshwater sediments spiked with endrin were tested using the amphipod Hyalella azteca; while the midge, Chironomus tentans, was tested using kepone-spiked sediments. Figure 4-1 presents the percentage mortalities of the benthic species tested in individual treatments for each chemical versus "pore water toxic units" (PWTUs) for all sediments tested. PWTUs are the concentration of the chemical in pore water (μg/L) divided by the water only LC50 (μg/L). Theoretically,

4-4

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

Common/Sci. Name	Method[a] Duration(days)	Water Only LC50 μg/L	Pore Water LC50 μg/L	TOC (%)	Sediment Acenaphthene LC50 mg/g Dry Wt.	μg/g OC[d]	Predicted[c] LC50 μg/g OC	Ratio: Actual LC50/Predicted LC50	Reference
Amphipod, Rhaustorius estuarius	FT,M/10	374	800	1.23 0.82[d]	44.4	4,330	2,152	2.01	Swartz, 1991
Amphipod, Rhaustorius estuarius	FT,M/10	374	609	2.49	47.8	1,920	2,152	0.89	Swartz, 1991
Amphipod, Rhaustorius estuarius	FT,M/10	374	542	4.21	68.4	1,630	2,152	0.76	Swartz, 1991
Amphipod, Leptocheirus plumulosus	FT,M/10	678	>1,720	1.62 0.82[d]	>193	>23,500	3,900	>6.02	Swartz, 1991
Amphipod, Leptocheirus plumulosus	FT,M/10	678	1,410	2.52	193	7,730	3,900	1.98	Swartz, 1991
Amphipod, Leptocheirus plumulosus	FT,M/10	678	1,490	3.66 2.97[d]	382	11,200	3,900	2.87	Swartz, 1991

[a]FT = flow-through, M = measured concentration

[b]Treatment-specific TOC concentrations were used in LC50 calculations.

[c]Predicted LC50 ($\mu g/g_{oc}$) = water-only LC50 ($\mu g/L$) x K_{oc} (L/Kg_{oc}) x 1 $Kg_{oc}/1000g_{oc}$; Where K_{oc} = $10^{3.76}$

[d]TOC of sediment used in highest concentration(s).

4-5
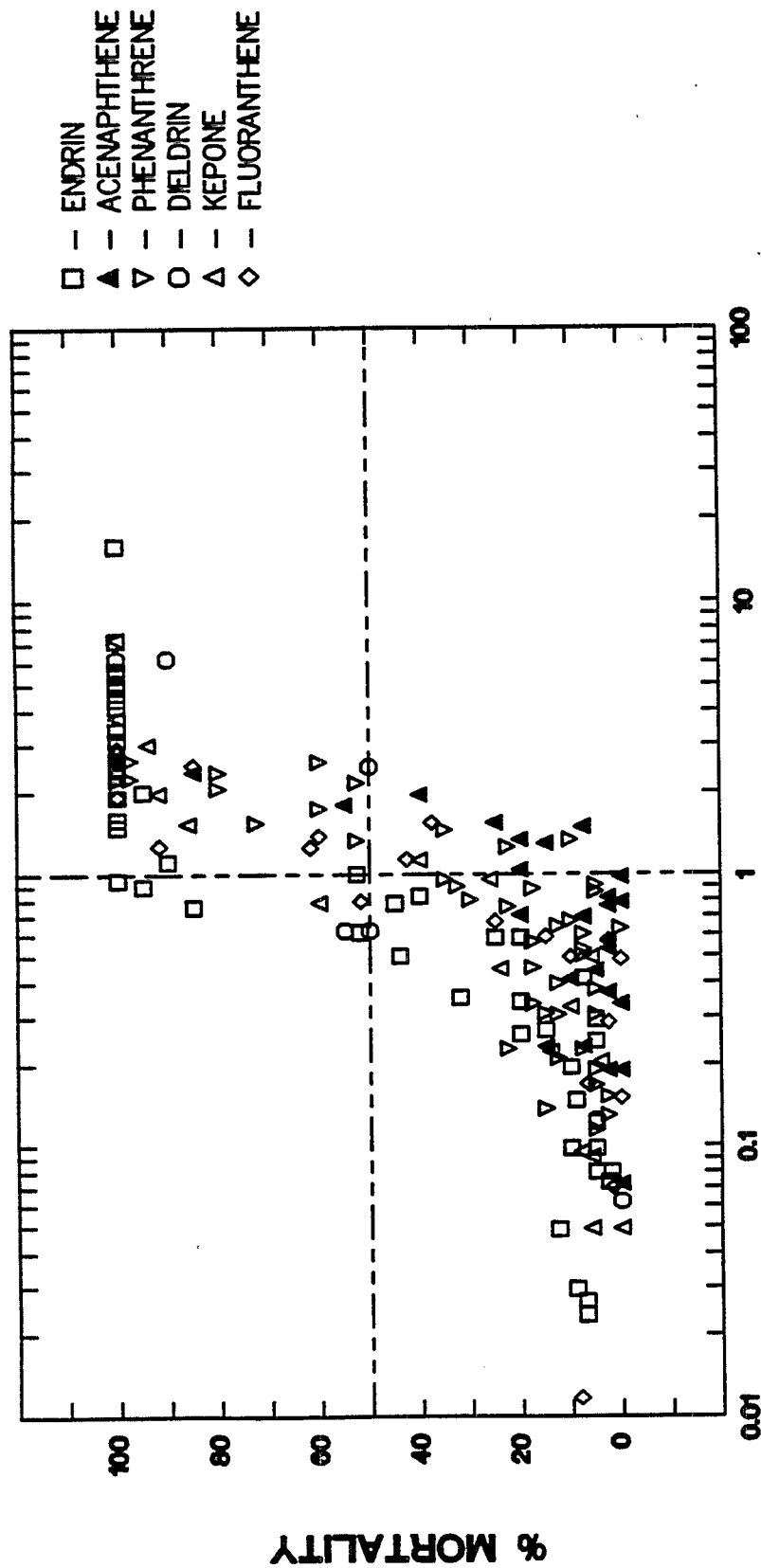
PORE WATER TOXIC UNITS

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, dieldrin fluoranthene and phenanthrene SQC documents, and original references for raw data.)

4-6

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. Data from the dieldrin experiments (Hoke and Ankley, 1991) are not included because more knowledge of the pore water DOC will be required because dieldrin has a high K_{OC} value.

4.3 TESTS OF THE EQUILIBRIUM PARTITIONING PREDICTION OF SEDIMENT TOXICITY:

SQC derived using the EqP approach utilize partition coefficients (K_{oc}) and FCV from WQC documents to derive the SQC concentration for protection of benthic organisms. The 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 acenaphthene in sediments are available for four benthic species. Data from tests with water column species were not included in this analysis. Testing of this component of SQC derivation requires three elements: (1) a water-only effect concentration, such as a 10-day LC50 value in μg/L, (2) an identical sediment effect concentration on an organic carbon basis, such as a 10-day LC50 value in μg/g$_{oc}$, and (3) a partition coefficient for the chemical, K_{oc} in L/Kg$_{oc}$. This

4-7

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 10-day LC50 values from acenaphthene-spiked sediment tests on a $\mu g/g_{oc}$ basis with B. estuarius and L. plumulosus were calculated (Table 4-2) using the $\log_{10}K_{oc}$ value of 3.78 from Section 2 of this document and the sediment LC50s in Swartz (1991). Ratios of actual to predicted LC50s for acenaphthene averaged 1.11 (range 0.76 to 2.01) for E. estuarius and 3.25 (range 1.98 to >6.02) for L. plumulosus. The overall mean for both species was 190.

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} obtained using the slow-stir flask method is not available. Swartz (1991) exposed the saltwater amphipods E. estuarius and L. plumulosus to acenaphthene in three marine sediments having organic carbon contents ranging from 0.82 to 4.2% and to phenanthrene in three marine sediments having organic carbon contents ranging from 0.82 to 3.6%.. Swartz et al. (1990) exposed the saltwater amphipod R. abronius to fluoranthene in three marine sediments having 0.18, 0.31 and 0.48% organic carbon. Hoke and Ankley (1991) exposed the amphipod H. azteca to three dieldrin-spiked freshwater sediments having 1.7, 3.0 and 8.5% organic carbon and Hoke (1992) exposed the midge C. tentans to 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-

4-8

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.

PREDICTED SEDIMENT TOXIC UNIT

Legend:
- □ — ENDRIN
- ▲ — ACENAPHTHENE
- ▽ — PHENANTHRENE
- ○ — DIELDRIN
- ◇ — FLUORANTHENE
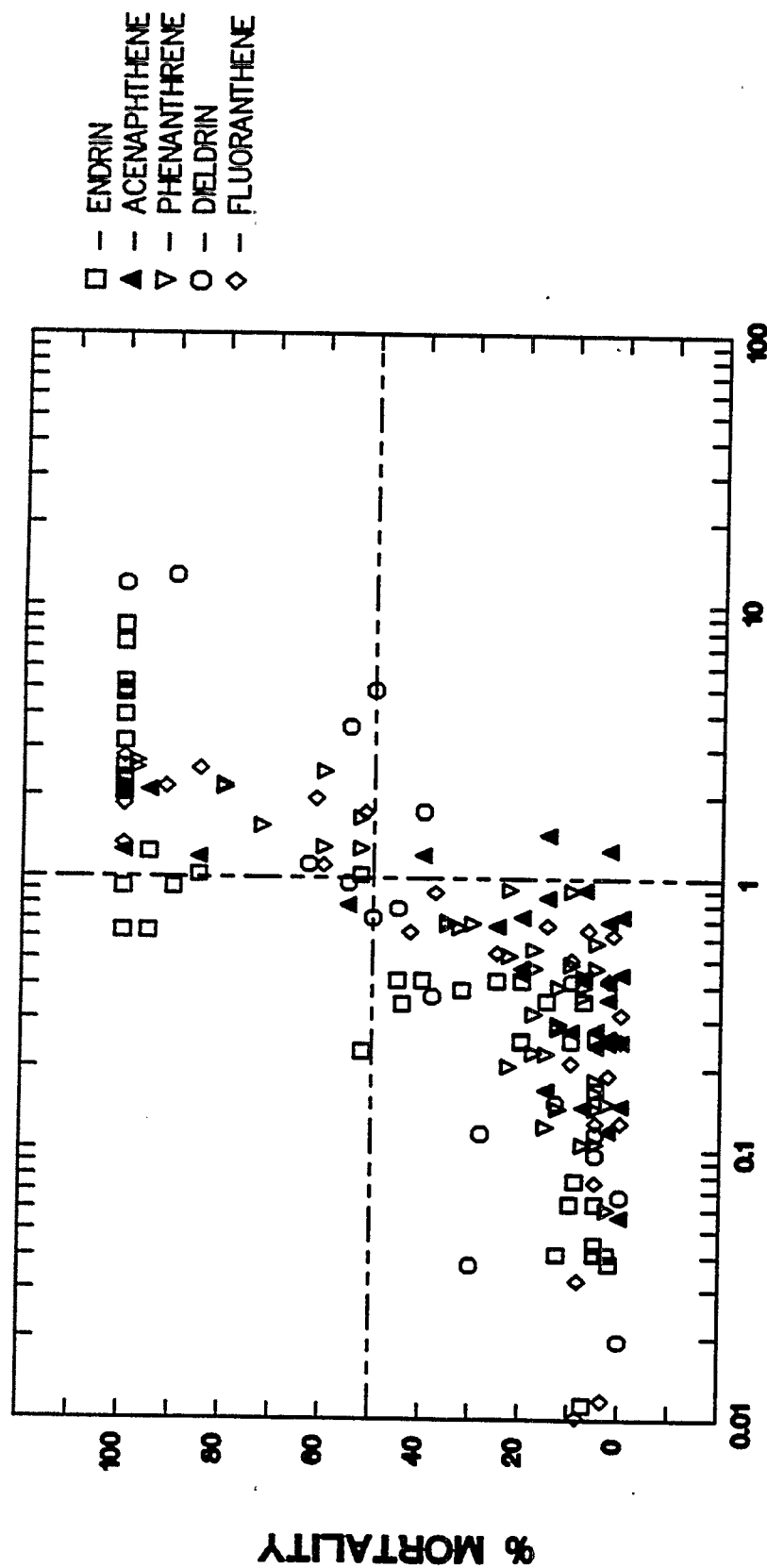
Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991), dieldrin (Hoke and Ankley, 1991), endrin (Nebeker et al., 1989; Schuytema et al., 1989) or fluoranthene (Swartz et al., 1990; De Witt et al., 1992) and midge in dieldrin spiked sediments (Hoke, 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 ($K_{OC} \times$ Water-only LC50, $\mu g/L$). (See Appendix B in this document and Appendix B in the dieldrin, endrin, fluoranthene, and phenanthrene SQC documents for raw data).

4-10

SECTION 5

CRITERIA DERIVATION FOR ACENAPHTHENE

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 the calculate the sediment quality criteria (SQC) because it is probable that the concentration of contaminants in sediments are relatively stable over time, thus exposure to sedentary benthic species should be chronic and relatively constant. This is in contrast to the situation in the water column, where a rapid change in exposure and exposures of limited durations can occur due to fluctuations in effluent concentrations dilutions in receiving waters or the free-swimming or planktonic nature of water column organisms. For some particular uses of the SQC it may be appropriate to use the areal extent and vertical stratification of contamination of a sediment at a site in much the same way that averaging periods are mixing zones used 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 acenaphthene 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:

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

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, SQC_{oc} (μg/g$_{oc}$), is:

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

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 acenaphthene 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

5-2

SQC may not apply.

TABLE 5-1. SEDIMENT QUALITY CRITERIA FOR ACENAPHTHENE.

Type of Water Body	$\text{Log}_{10}\text{K}_{\text{ow}}$ (L/kg)	$\text{Log}_{10}\text{K}_{\text{oc}}$ (L/kg)	FCV (μg/L)	SQC_{OC} (μg/g_{oc})
Fresh Water	3.83	3.76	23.0	130
Salt Water	3.83	3.76	40.4	230

[a] $\text{SQC}_{\text{OC}} = (10^{3.76} \text{ L/kg}_{\text{oc}}) \bullet (10^{-3} \text{ kg}_{\text{oc}}/\text{g}_{\text{oc}}) \bullet (23.0 \ \mu\text{g acenaphthene/L}) = 130 \ \mu\text{g acenaphthene/g}_{\text{oc}}$

[b] $\text{SQC}_{\text{OC}} = (10^{3.76} \text{ L/kg}_{\text{oc}}) \bullet (10^{-3} \text{ kg}_{\text{oc}}/\text{g}_{\text{oc}}) \bullet (40.4 \ \mu\text{g acenaphthene/L}) = 230 \ \mu\text{g acenaphthene/g}_{\text{oc}}$

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:

$$\mu\text{g Chemical/g}_{\text{oc}} = \mu\text{g Chemical/g}_{\text{DRY WT}} \div (\% \text{ TOC} \div 100)$$

$$= \mu\text{g Chemical/g}_{\text{DRY WT}} \bullet 100 \div \% \text{ TOC}$$

For example, a freshwater sediment with a concentration of 6.00 μg chemical/$\text{g}_{\text{DRY WT}}$

and 0.5% TOC has an organic carbon-normalized concentration of 1,200 $\mu g/g_{OC}$ (6.00 $\mu g/g_{DRYWT}$ • 100 ÷ 0.5 = 1,200 $\mu g/g_{OC}$) which exceeds the SQC of 130 $\mu g/g_{OC}$. Another freshwater sediment with the same concentration of acenaphthene (6.00 $\mu g/g_{DRY\ WT}$) but a TOC concentration of 5.0% would have an organic carbon normalized concentration of 120 $\mu g/g_{OC}$ (6.00 $\mu g/g_{DRY\ WT}$ • 100 ÷ 5.0 = 120 $\mu g/g_{OC}$), which is below the SQC for acenaphthene.

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 1.3 $\mu g/g$:

$$SQC_{DRY\ WT.} = 130\ \mu g/g_{OC} \bullet 1\%\ TOC \div 100 = 1.3\ \mu g/g_{DRY\ WT}$$

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 acenaphthene 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 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 acenaphthene 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:

$$LC50_{s,oc} = K_{oc}LC50_w \qquad (5-3)$$

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 ϵ 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 is:

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

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 mg/L. The statistical problem is to estimate μ_i, σ_α^2, and σ_α^2 The maximum likelihood method is used to

make these estimates (U.S. EPA, 1993a). The results are shown in Table 5-2.

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

Source of Uncertainty	Parameter $(\mu g/g_{oc})$	Value
Exposure media	σ_α	0.39
Replication	$\sigma_\in$	0.21
Sediment Quality Criteria	$\sigma_{SQC}{}^a$	0.39

$^a\sigma_{SQC} = \sigma_\alpha$

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

The confidence limits for the 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:

$$\ln(SQC_{oc})_{UPPER} = \ln(SQC_{oc}) + 1.96\sigma_{SQC} \tag{5-5}$$

$$\ln(SQC_{oc})_{LOWER} = \ln(SQC_{oc}) - 1.96\sigma_{SQC} \tag{5-6}$$

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 does not

apply and the SQC do not apply.

TABLE 5-3. SEDIMENT QUALITY CRITERIA CONFIDENCE LIMITS FOR ACENAPHTHENE
Sediment Quality Criteria

Type of Water Body	SQC_{OC} $\mu g/g_{OC}$	95% Confidence Limits ($\mu g/g_{OC}$)	
		Lower	Upper
Fresh Water	130	62	280
Salt Water	240	110	500

5.3 COMPARISON OF ACENAPHTHENE 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 determined from effect concentrations from toxicity tests with benthic species exposed to sediments spiked with acenaphthene and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figure 5-1; 5-2). Effect concentrations in sediments can be predicted from water-only toxicity data and K_{OC} values (See Section IV). Acute-chronic ratios (ACRs) are used to extrapolate from Genus Mean Acute Values (GMAV) from water-only 96-hour lethality tests to chronically acceptable concentrations. Therefore, it may be reasonable to predict for acenaphthene chronically acceptable sediment concentrations (Predicted Genus Mean Chronic Value (PGMCV)) from GMAVs (Appendix A), ACRs (Table 3-3) and the K_{OC} (Table 5-1):

$$PGMCV = (GMAV \div ACR) \bullet K_{OC}$$

5-8

Figure 5-1. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to acenaphthene-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, O for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to acenaphthene-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 $\triangle$ for arthropods, $\bigcirc$ for fishes and $\square$ 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.

Each predicted PGMCV for tested fishes ● , arthropods ▲ or other invertebrates ■ is plotted against the percentage rank of its sensitivity. Results from toxicity tests with benthic organisms exposed to sediments spiked with acenaphthene (Table 4-1) are placed in the PGMCV rank appropriate to the test-specific effect concentration. (For example, the 10-day LC50 for E. estuarius (2,384 $\mu g/g_{oc}$) is placed between the 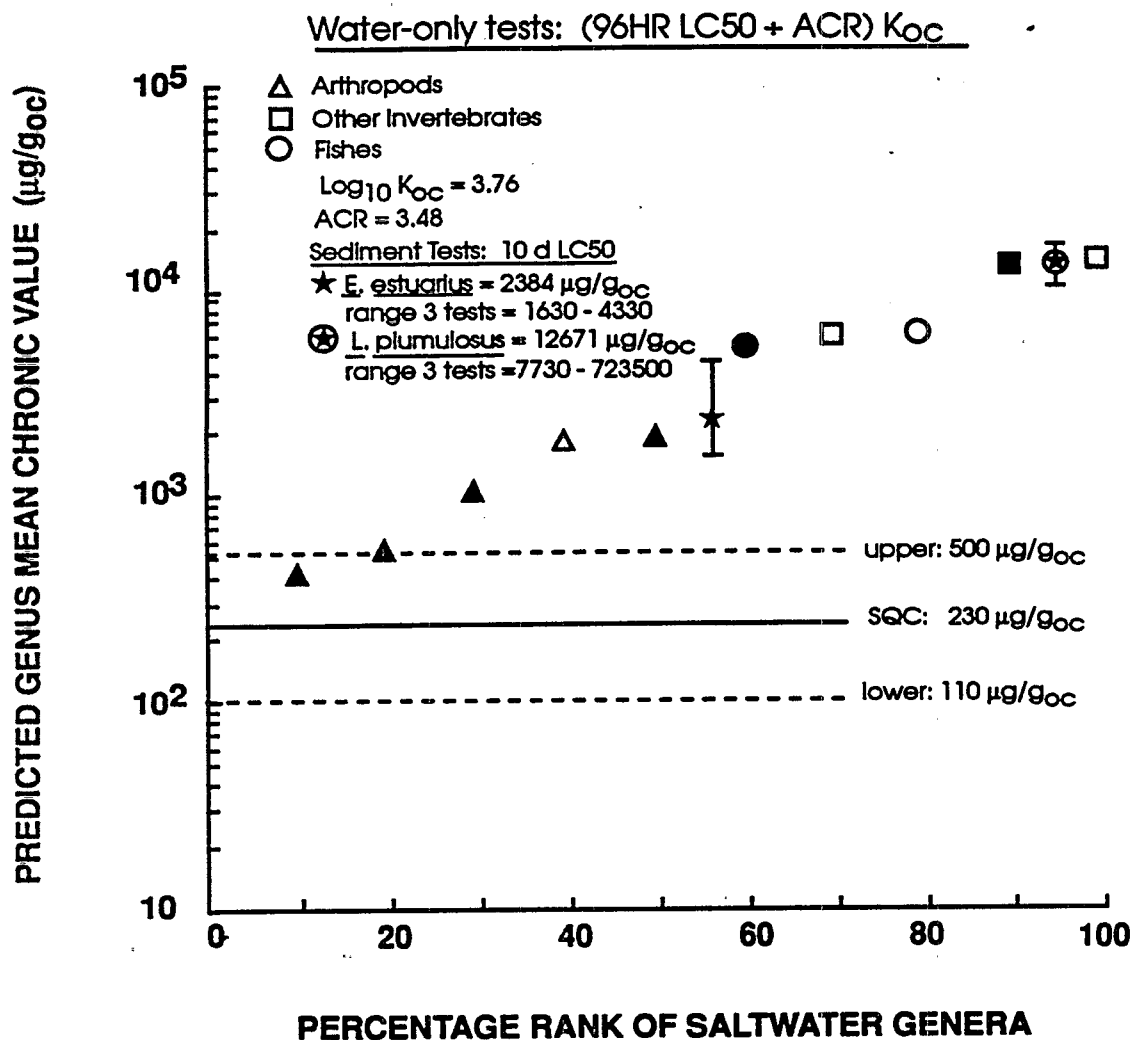PGMCV of 1,858 $\mu g/g_{oc}$ for the amphipod, Leptocheirus, and the PGMCV of 5,120 $\mu g/g_{oc}$ for the minnow, Cyprinodon.) Therefore, LC50 or other effect concentrations are intermingled in this figure with concentrations predicted to be chronically safe. Care should be taken by the reader in interpreting 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 acenaphthene 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 acenaphthene.

The freshwater SQC for acenaphthene (130 $\mu g/g_{oc}$) is less than any of the PGMCVs. PGMCVs for 9 of 10 freshwater genera are greater than the upper 95% confidence interval of the SQC (280 $\mu g/g_{oc}$). The PGMCVs for the cladoceran Daphnia (198 $\mu g/g_{oc}$) is below the SQC upper 95% confidence interval. This suggests that if the extrapolation from water only acute lethality tests to chronically acceptable sediment concentrations is accurate, this 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 acenaphthene, the PGMCVs range over an order of magnitude from the most sensitive to the most tolerant

5-11

genus. Chronic effect concentrations may, however, occur at concentrations below saturation. A sediment concentration four times the SQC would include the PGMCVs of two of the three benthic genera tested including stoneflies, and amphipods. The tolerant benthic midge Paratanytarsus might be expected to not be chronically impacted in sediments with acenaphthene concentrations 25X the SQC.

The saltwater SQC for acenaphthene (230 $\mu g/g_{OC}$) is less than any of the PGMCVs or LC50 values from spiked sediment toxicity tests. The PGMCV for the sand shrimp Crangon septemspinosus (404 $\mu g/g_{OC}$) is lower than the upper 95% confidence interval for the SQC. For acenaphthene, PGMCVs from the most sensitive to the most tolerant saltwater genus range over an order of magnitude. A sediment concentration five times the SQC would include the PGMCVs of one-half of the six benthic genera tested including three arthropod genera. Other genera of benthic arthropods, polychaetes, and fishes are less sensitive. Data from lethality tests with two saltwater amphipods, Eohaustorius and Leptocheirus, exposed to acenaphthene spiked into sediments substantiates this projection; the 10 day LC50s from these tests range from 10 to 40 times the SQC of 230 $\mu g/g_{OC}$.

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

A STORET (U.S. EPA, 1989a) data retrieval was performed to obtain a preliminary assessment of the concentrations of acenaphthene in the sediments of the nation's water bodies. Log probability plots of acenaphthene concentrations on a dry weight basis in sediments since 1986 are shown in Figure 5-3. Acenaphthene is found at detectable concentrations in sediments from rivers, lakes and near coastal water bodies in the United States. Median concentrations are
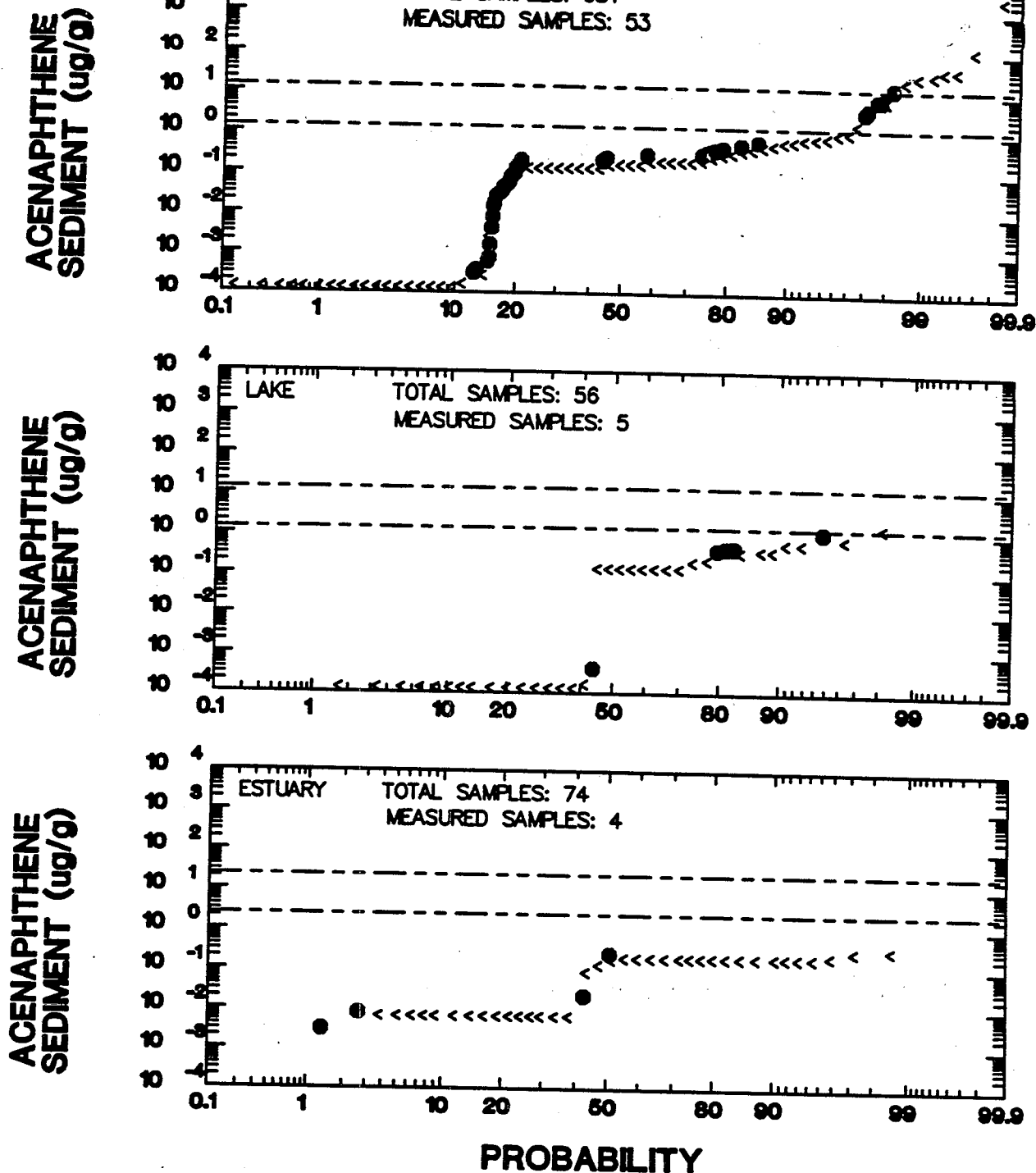
Figure 5-3. Probability distribution of concentrations of acenaphthene in sediments from streams, lakes and estuaries in the United States from 1986 to 1990 from the STORET (U.S. EPA, 1989b) database, compared to the acenaphthene SQC values of 13 $\mu g/g$ in freshwater sediments having TOC = 10% and 1.3 $\mu g/g$ in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 23 $\mu g/g$ when TOC =10% and 2.3 $\mu g/g$ when TOC=1%. The upper dashed line on each figure represents the SQC value when TOC = 10%, the lower dashed line represents the SQC when TOC = 1%.

5-13

generally at about 0.1 μg/g in each of these three types of water bodies. Acenaphthene concentrations in sediments range over seven orders of magnitude throughout the country.

The SQC for acenaphthene can be compared to existing concentrations of acenaphthene 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 percent are compared to acenaphthene's distribution in sediments as examples only. For fresh water sediments, SQC values are 1.3 μg/g in sediments having 1% organic carbon and 13 μg/g dry weight in sediments having 10% organic carbon; for marine sediments SQC are 2.3 μg/g and 23 μg/g, respectively. Figure 5-3 presents the comparisons of these SQC to probability distributions of observed sediment acenaphthene levels for streams and lakes (fresh water systems, shown on the upper panels) and estuaries (marine systems, lower panel). For streams (n = 681) the SQC of 1.3 μg/g for 1 % organic carbon fresh water sediments is exceeded by less than 4 % of the data; the 13 μg/g criteria for 10% organic carbon freshwater sediments is exceeded in about 2 % of the samples. For lakes (n = 56), the SQC for 1 % organic carbon fresh water sediments is exceeded by about 2% of the samples, while the SQC for 10 % organic carbon fresh water sediments is not exceeded by any of the lake samples. In estuaries, the data (n = 74) indicate that neither of the criteria of 2.3 μg/g dry weight for salt water sediments having 1 % organic carbon or 23 μg/g dry weight for salt water sediments having 10 % organic carbon are exceeded, although the STORET database for marine sediments is not as extensive as the database for freshwater sediments.

The acenaphthene distribution in Figure 5-3 includes data from some samples in which

the acenaphthene 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 acenaphthene (230 $\mu g/g_{oc}$) on Figure 5-4. Data presented are from sediments with 0.20 to 31.9% organic carbon. None of these samples (n=288) exceeded the criteria. Hence, these results are consistent with the preceding comparison of the marine SQC to STORET data.

Regional differences in acenaphthene 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 areas. It is presented as an aid in assessing the range of reported acenaphthene sediment concentrations and the extent to which they may exceed the sediment quality criteria.

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
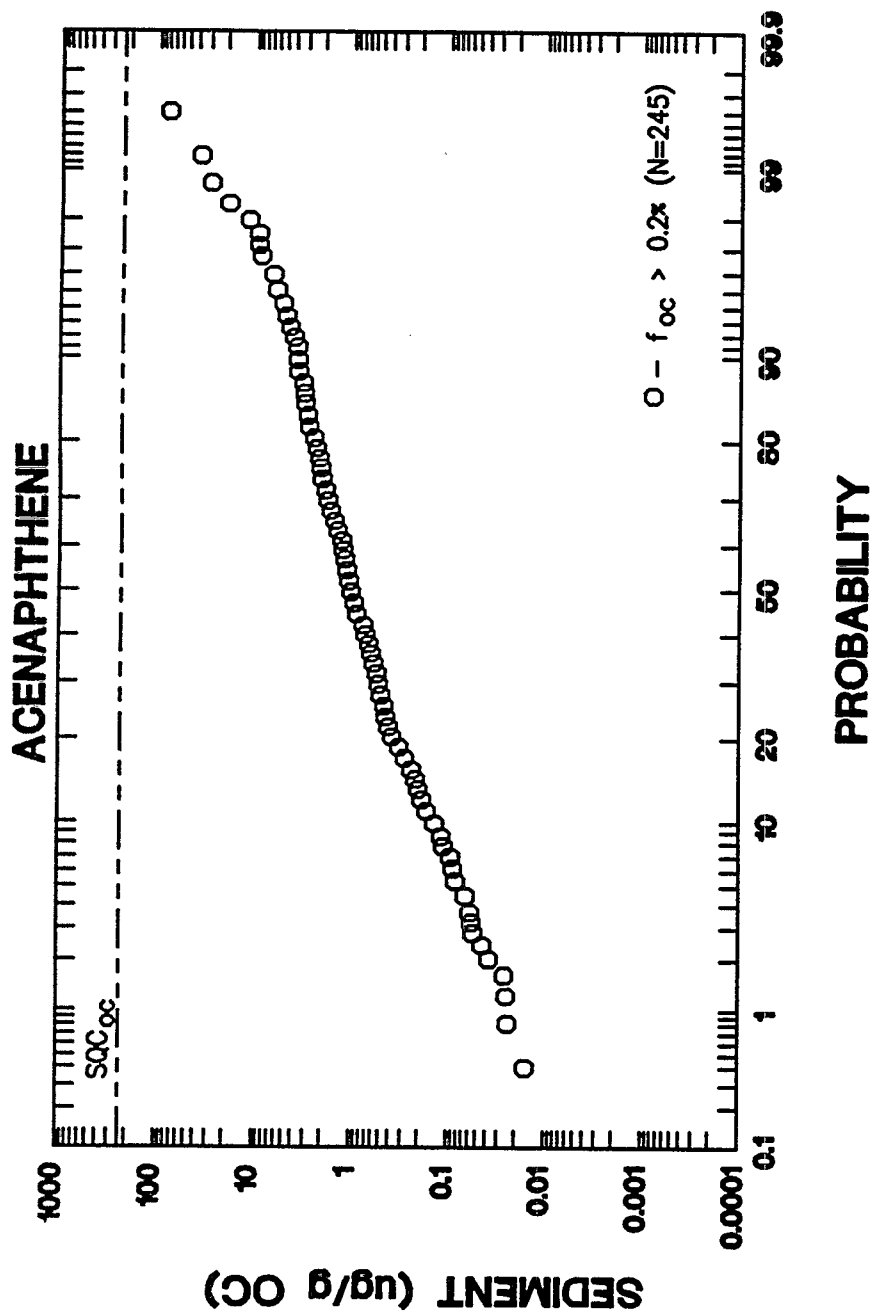
Figure 5-4.   Probability distribution of concentrations of acenaphthene in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal line is the saltwater SQC value of 230 $\mu g/g_{OC}$.

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.

The concerns about mixtures of contaminants are particularly important with the PAHs, which almost invariably occur as complex mixtures. Some guidance on interpretations of PAH concentrations is possible given the presence of SQC for acenaphthene and other individual PAHs. This is because much is known about the toxicity and structure-activity relationships of the so-called narcosis chemicals, a group of nonionic organic chemicals to which the PAHs belong. The toxicity of the narcosis chemicals is additive (Broderius and Kahl, 1985). The toxicity of these chemicals increases with increasing K_{ow} (Veith et al., 1983) and their bioavailability in sediments decreases as a function of its K_{ow}. Therefore, the toxicities of many PAHs in sediments are likely to be similar. This explains why SQC values for fluoranthene (fresh: 620 $\mu g/g_{oc}$, salt: 300 $\mu g/g_{oc}$), acenaphthene (fresh: 130 $\mu g/g_{oc}$, salt: 230 $\mu g/g_{oc}$) and phenanthrene (fresh: 180 $\mu g/g_{oc}$, salt: 240 $\mu g/g_{oc}$) differ little and why it is theoretically possible to develop an SQC for total PAHs. EPA is currently conducting research aimed at development of SQC for combined PAHs.

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, antagonisms have 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 like cinder, soot, or oil droplets contain PAHs, disequilibrium exists and criteria are over protective. In liquid chemical spill situations disequilibrium concentrations in interstitial and overlying water may be proportionately higher relative to sediment concentrations. In this case criteria may be underprotective.

In very dynamic areas, with highly erosional or depositional bedded sediments, equilibrium may not be attained with contaminants. However, even high K_{ow} nonionic organic compounds come to equilibrium in clean sediment in a period of days, weeks or months. Equilibrium times are shorter for mixtures of two sediments each previously at equilibrium. This is particularly relevant in tidal situations where large volumes of sediments are eroded and deposited, yet near equilibrium conditions may predominate over large areas. Except for spills and particulate chemical, near equilibrium is the rule and disequilibrium is uncommon. In instances where it is suspected that EqP does not apply for a particular sediment because of

disequilibrium discussed above, site-specific methodologies may be applied (U.S. EPA, 1993b).

SECTION 6

CRITERIA STATEMENT

The procedures described in the "Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants by Using Equilibrium Partitioning" (U.S. EPA, 1993a) indicate that benthic organisms should be acceptably protected in freshwater sediments containing $\leq$ 130 μg acenaphthene/g organic carbon and saltwater sediments containing $\leq$ 230 μg acenaphthene/g organic carbon, except possibly where a locally important species is very sensitive or sediment organic carbon is $< 0.2\%$.

Confidence limits of 62 to 280 $\mu g/g_{oc}$ for freshwater sediments and 110 to 500 $\mu g/g_{oc}$ 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 acenaphthene, 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 acenaphthene in sediments that would be protective of benthic species. It is the philosophy of the Agency and the EPA Science Advisory Board that the use of sediment quality criteria (SQCs) as stand-alone, pass-fail criteria is not recommended for all applications and should

frequently trigger additional studies at sites under investigation. The upper confidence limit should be interpreted as a concentration above which impacts on benthic species should be expected. Conversely, the lower confidence limit should be interpreted as a concentration below which impacts on benthic species should be unlikely.

SECTION 7

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

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-[d] TRATION	LC50/[e] EC50 μg/L	HMAV SPECIES[f] μg/L	HMAV GENUS[g] μg/L	OVERALL[h] GMAV μg/L	REFERENCES[i]
FRESHWATER SPECIES									
Snail, Aplexa hypnorum	X	W	F	M	> 2,040	>2,040	>2,040	>2,040	Holcombe et al., 1983
Cladoceran, Daphnia magna	X	W	S	M	320	-	-	-	EG&G, Bionomics, 1982
Cladoceran, Daphnia magna	X	W	S	M	1,300	-	-	-	EG&G, Bionomics, 1982
Cladoceran, Daphnia magna	X	W	F	M	120	-	-	-	EG&G, Bionomics, 1982
Cladoceran, Daphnia magna	X	W	S	U	41,000	-	-	-	LeBlanc, 1980
Cladoceran, Daphnia magna	X	W	S	U	3,450	120	120	120	Randall and Knopp, 1980
Amphipod, Gammarus minus	X	E	S	U	460	460	460	460	Horne et al., 1983
Stonefly, Peltoperla maria	X	E	S	U	240	240	240	240	Horne et al., 1983
Midge, Paratanytarsus sp.	X	E	S	M	2,000	-	-	-	Northwestern Aquatic Science, Inc., 1982
Midge, Paratanytarsus sp.	X	E	S	M	2,090	2,045	2,045	2,045	Northwestern Aquatic Science, Inc., 1982
Rainbow trout, Oncorhynchus mykiss	J	W	F	M	670	670	670	670	Holcombe et al., 1983
Brown trout, Salmo trutta	J	W	F	M	580	580	580	580	Holcombe et al., 1983

A-1

Appendix A. - Acenaphthene: (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-[d] TRATION	LC50/[e] EC50	HMAV SPECIES[f]	HMAV GENUS[g]	OVERALL[h] GMAV	REFERENCES[i]
					μg/L	μg/L	μg/L	μg/L	
Fathead minnow, Pimephales promelas	J	W	F	M	608	-	-	-	Cairns and Nebeker, 1982
Fathead minnow, Pimephales promelas	J	W	F	M	>1,400	-	-	-	EG&G, Bionomics, 1982
Fathead minnow, Pimephales promelas	J	W	S	M	1,500	-	-	-	EG&G, Bionomics, 1982
Fathead minnow, Pimephales promelas	J	W	F	M	1,600	-	-	-	Holcombe et al., 1983
Fathead minnow, Pimephales promelas	A	W	R	U	3,700	-	-	-	Academy of Natural Science, 1981
Fathead minnow, Pimephales promelas	J	W	S	M	3,100	986	986	986	Marine Bioassay Laboratories, 1981
Channel catfish, Ictalurus punctatus	J	W	F	M	1,720	1,720	1,720	1,720	Holcombe et al., 1983
Bluegill, Lepomis macrochirus	J	W	S	U	1,700	1,700	1,700	1,700	Buccafusco et al., 1981; Ward et al., 1981

SALTWATER SPECIES

COMMON/SCI. NAME	LIFE STAGE	HAB-ITAT	METHOD	CONCEN-TRATION	LC50/EC50	HMAV SPECIES	HMAV GENUS	OVERALL GMAV	REFERENCES
Annelid worm, Neanthes arenaceodentata	J	I	R	U	16,440	-	-	-	Thursby et al., 1989a
Annelid worm, Neanthes arenaceodentata	X	I	S	U	3,600	7,693	7,693	7,693	Horne et al., 1983
Slipper limpet, Crepidula fornicata	L	W	R	U	3,436	3,436	3,436	3,436	Thursby et al., 1989a
Mysid, Mysidopsis bahia	J	E	R	U	1,190	-	-	-	Thursby et al., 1989a
Mysid, Mysidopsis bahia	J	E	S	U	970	-	-	-	U.S. EPA, 1978; Ward et al., 1981
Mysid, Mysidopsis bahia	J	E	F	M	460	-	-	-	Thursby et al., 1989b

Appendix A. - Acenaphthene: (continued)

COMMON/SCI. NAME	LIFE[a] STAGE	HAB-[b] ITAT	METHOD[c]	CONCEN-TRATION[d]	LC50/[e] EC50 (µg/L)	HMAV SPECIES[f] (µg/L)	HMAV GENUS[g] (µg/L)	OVERALL[h] GMAV (µg/L)	REFERENCES[i]
Mysid, Mysidopsis bahia	J	E	S	M	160	-	-	-	EG&G, Bionomics, 1982
Mysid, Mysidopsis bahia	J	E	F	M	190	-	-	-	EG&G, Bionomics, 1982
Mysid, Mysidopsis bahia	J	E	F	M	466.1	-	-	-	Horne et al., 1983; Thursby, 1991b
Mysid, Mysidopsis bahia	J	E	F	M	250	317.7	317.7	317.7	Horne et al., 1983
Amphipod, Ampelisca abdita	J	I	R	U	1,125	1,125	1,125	1,125	Thursby et al., 1989a
Amphipod, Leptocheirus plumulosus	A	I	F	M	589.4	589.4	589.4	589.4	Swartz et al., 1991
Grass shrimp, Palaemonetes pugio	L	W	R	U	1,697	-	-	-	Thursby et al., 1989a
Grass shrimp, Palaemonetes pugio	X	W	S	U	676.8	1,072	1,072	1,072	Horne et al., 1983; Thursby, 1991b
Sand shrimp, Crangon septemspinosus	X	E	S	U	245.0	245.0	245.0	245.0	Horne et al., 1983; Thursby, 1991b
Sea urchin, Arbacia punctalata	E	W	S	U	8,163	8,163	8,163	8,163	Thursby et al., 1989a
Sheepshead minnow, Cyprinodon variegatus	J	E,W	R	U	>50,000	-	-	-	Thursby et al., 1989a
Sheepshead minnow, Cyprinodon variegatus	A	E,W	F	M	3,100	-	-	-	Ward et al., 1981
Sheepshead minnow, Cyprinodon variegatus	J	E,W	S	U	2,200	3,100	3,100	3,100	Heitmuller et al., 1981
Inland silverside, Menidia beryllina	J	W	R	U	5,564	-	-	-	Thursby et al., 1989a
Inland silverside, Menidia beryllina	X	W	S	U	2,300	3,577	3,577	3,577	Horne et al., 1983

A-3

Appendix A. Acenaphthene: (continued)

[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 Acenaphthene Water Quality Criteria document (U.S. EPA, 1980) or in the references section of this Sediment Quality Criteria document.

A-4

SEDIMENT SOURCE/ SPECIES TESTED	MORTALITY (%)	SEDIMENT CONCENTRATION μg/g DRY WT.	ORG. CAR.	PORE WATER CONCENTRATION (ug/L)	TOC %	Log K_{oo} [a]	REFERENCES
South Beach, OR. Eohaustorius estuarius	7.5	3.8	309	87	1.23	3.55	Swartz, 1991
	10	7.2	585	154	1.23	3.58	
	20	11.7	951	264	1.23	3.56	
	20	19.0	1545	498	1.23	3.49	
	40	32.1	2610	732	1.23	3.55	
	77.5	66.2	8073	1060	0.82	3.88	
McKinney Slough, OR. Eohaustorius estuarius	5	13.0	522	165	2.49	3.50	Swartz, 1991
	2.5	22.1	888	306	2.49	3.46	
	25	35.7	1434	575	2.49	3.40	
	85	63.6	2554	888	2.49	3.46	
	95	111	4458	1133	2.49	3.59	
Eckman Slough Site 1, OR. Eohaustorius estuarius	2.5	23.5	558	199	4.21	3.45	Swartz, 1991
	20	41.9	995	387	4.21	3.41	
	55	71.6	1701	670	4.21	3.40	
	100	113	2684	976	4.21	3.44	
	100	183	4347	1195	4.21	3.56	
South Beach, OR. plumulosus Leptocheirus	0	3.6	222	49	1.62	3.65	Swartz, 1991
	2.5	7.5	463	129	1.62	3.55	
	0	45.8	2827	659	1.62	3.63	
	7.5	57.5	3549	1002	1.62	3.55	
	2.5	121	14756	1470	0.82	4.00	
	20	193	23537	1720	0.82	4.14	
McKinney Slough, OR. Leptocheirus plumulosus	0	14.3	567	129	2.52	3.64	Swartz, 1991
	2.5	24.3	964	249	2.52	3.59	
	7.5	42.2	1675	470	2.52	3.55	
	2.5	67.5	2679	516	2.52	3.72	
	2.5	124.1	4925	1140	2.52	3.64	
	60	210	8333	1470	2.52	3.75	
	75	422	16746	1500	2.36	4.05	

APPENDIX B. Acenaphthene (cont'd)

SEDIMENT SOURCE/ SPECIES TESTED	MORTALITY (%)	SEDIMENT CONCENTRATION, μg/g DRY WT.	ORG. CAR.	PORE WATER CONCENTRATION (ug/L)	TOC %	Log K_{oc}^*	REFERENCES
Eckman	15	23.9	653	158	3.66	3.61	Swartz, 1991
Site 1, OR.	5	39.7	1085	299	3.66	3.56	
Leptocheirus	0	63.3	1730	533	3.66	3.51	
plumulosus	15	121	3306	874	3.66	3.58	
	15	206	5628	1210	3.66	3.67	
	45	356	9727	1490	3.66	3.81	
	77.5	501	16869	1580	2.97	4.03	

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

B-2