

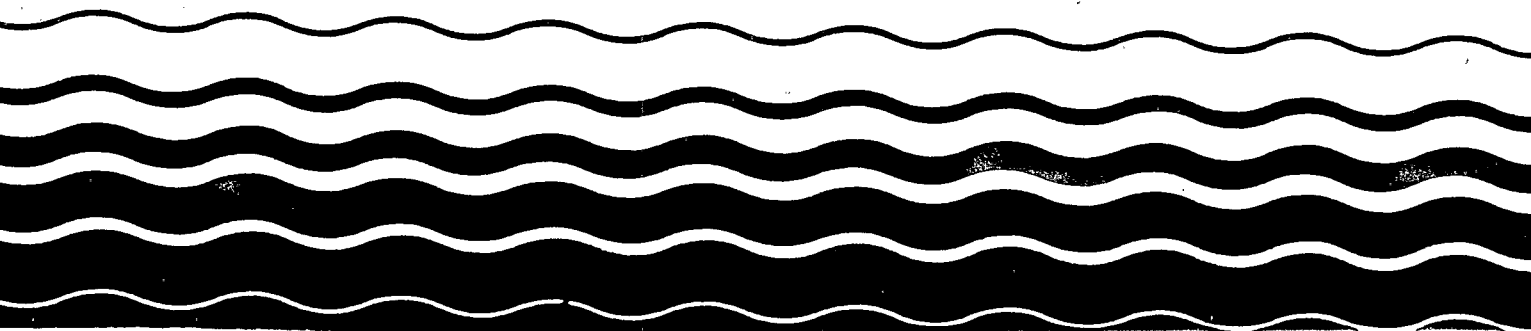


Sediment Quality Criteria for the Protection of Benthic Organisms:

FLUORANTHENE



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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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Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991a), 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 ($\mu\text{g/g}_{\text{OC}}$) divided by the predicted LC50 ($\mu\text{g/g}_{\text{OC}}$) in sediments ($K_{\text{OC}} \times \text{Water-only LC50 } (\mu\text{g/L}) \times 1 \text{ Kg}_{\text{OC}}/1,000\text{g}_{\text{OC}}$). (See Appendix B in this document and Appendix B in the dieldrin, endrin, acenaphthene, and phenanthrene SQC documents for raw data).

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Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to fluoranthene-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. $\text{PGMCV} = (\text{GMAV} \div \text{ACR})K_{\text{OC}}$. Symbols for PGMCVs are Δ for arthropods, \circ 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.

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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, the 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 concentrations in pore water.
3. The distribution of sensitivities of benthic and water column organisms to chemicals are similar; thus, the currently established WQC final chronic values (FCV) can be used to define the acceptable effects concentration of a chemical freely-dissolved in pore water.

The EqP approach, therefore, assumes that: (1) the partitioning of the chemical between sediment organic carbon and interstitial water is at equilibrium; (2) the concentration in either

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

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

SQC generated using the EqP method are suitable for use in providing guidance to regulatory agencies because they are:

1. numerical values;
2. chemical specific;
3. applicable to most sediments;
4. predictive of biological effects; and

5. protective of benthic organisms.

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

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

SQC values may also need to be adjusted because of site specific considerations. For example, spill situations, in which the chemical equilibrium between water and sediments has not yet been reached, a sediment chemical concentration that is 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, or 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 Deriving Site-specific Sediment Quality Criteria for the Protection of Benthic Organisms" (U.S. EPA, 1993b). The SQC may be underprotective where the toxicity of other chemicals are additive with the SQC chemical or species of unusual sensitivity occur at the site.

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

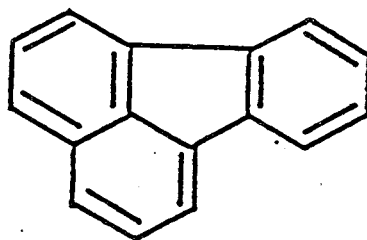
1.2 GENERAL INFORMATION: FLUORANTHENE

Fluoranthene is a member of the polycyclic aromatic hydrocarbon (PAH) group of organic compounds. Some sources of fluoranthene are crude oil, coal tar and motor oil (Verschuere, 1983). Fluoranthene is also produced naturally by plants, algae and bacteria (Suedel, 1989).

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).

Fluoranthene has a three ring structure and exists as pale yellow plates or needles (Figure 1-1). It has a solubility in water of 0.26 mg/L at 25°C and is a solid at room temperature (melting point of 111°C) (Verschueren, 1983). Two significant processes which can influence the fate of fluoranthene in the sediment are sorption and biodegradation (U.S. EPA, 1980). Sorption of fluoranthene onto solids in the water column and subsequent settling, as well as partitioning onto organics in the sediment, can significantly affect fluoranthene 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 fluoranthene in the sediment are oxidation, hydrolysis and volatilization (U.S. EPA, 1980).

In the absence of ultraviolet (UV) light the acute toxicity of fluoranthene ranges from 36 to >212 µg/L for freshwater and 52 to >20,000 µg/L for saltwater organisms (Appendix A). Differences between fluoranthene concentrations causing acute lethality and chronic toxicity in the absence of UV light in invertebrates are small; acute-chronic ratios (ACR) range from 3.385 to 3.404 for two species. The ACR for one species of fish is much larger (14.09). Sunlight photoactivates fluoranthene to increase its toxicity to up to over an order of magnitude greater than non-UV activated chemical (Appendix A). Although fluoranthene bioaccumulates in aquatic biota, the associated health or ecological risks are unknown.



MOLECULAR FORMULA
MOLECULAR WEIGHT
DENSITY
MELTING POINT
PHYSICAL FORM
VAPOR PRESSURE

$C_{16}H_{10}$
202.26
1.252 g/cc (25°C)
111°C
Pale yellow plates or needles
 5.0×10^{-6} mm Hg at 25°C

CAS NUMBER:
TSL NUMBER
CHEMICAL NAME:

206-44-0
LL 40250
Fluoranthene (Idryl,
Benzo[j,k]fluorene)

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

1.3 OVERVIEW OF DOCUMENT:

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

Bioavailability of a chemical at a particular sediment concentration often differs from one sediment type to another. Therefore, a method is necessary for determining a SQC based on the bioavailable chemical fraction in a sediment. For nonionic organic chemicals, the concentration-response relationship for the biological effect of concern can most often be correlated with the interstitial water (i.e., pore water) concentration (μg chemical/liter pore water) and not to the sediment chemical concentration (μg chemical/g sediment) (Di Toro et al.,

1991). From a purely practical point of view, this correlation suggests that if it were possible to measure the pore water chemical concentration, or predict it from the total sediment concentration and the relevant sediment properties, then that concentration could be used to quantify the exposure concentration for an organism. Thus, knowledge of the partitioning of chemicals between the solid and liquid phases in a sediment is a necessary component for establishing SQC. It is for this reason that the methodology described below is called the equilibrium partitioning (EqP) method.

It is shown in the SQC Technical Basis Document (U.S. EPA, 1993a) that the final acute values (FAVs) in the WQC documents are appropriate for benthic species for a wide range of chemicals. (The data showing this for fluoranthene 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 ($\mu\text{g/L}$) be the acceptable concentration in water for the chemical of interest; then compute the SQC using the partition coefficient, (K_p ; $\text{L/kg}_{\text{sediment}}$), between sediment and water:

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

This is the fundamental equation used to generate the SQC. Its utility depends upon the existence of a methodology for quantifying the partition coefficient, K_p .

Organic carbon appears to be the dominant sorption phase for nonionic organic chemicals

in naturally occurring sediments and thus controls the bioavailability of these compounds in sediments. Evidence for this can be found in numerous toxicity tests, bioaccumulation studies and chemical analyses of pore water and sediments (Di Toro et al., 1991). The evidence for fluoranthene is discussed in this section and section 4. The organic carbon binding of a chemical in sediment is a function of that chemical's organic carbon partition coefficient (K_{oc}) and the weight fraction of organic carbon in the sediment (f_{oc}). The relationship is as follows:

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

It follows that:

$$SQ_{C_{oc}} = K_{oc} FCV \quad (2-3)$$

where $SQ_{C_{oc}}$ 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 fluoranthene.

2.2 DETERMINATION OF K_{ow} FOR FLUORANTHENE:

Several approaches have been used to determine K_{ow} for the derivation of SQC, as discussed in the SQC Technical Basis Document. At the U.S. EPA, Environmental Research Laboratory at Athens, GA (ERL,A) three methods were selected for measurement and two for estimation of K_{ow} . The measurement methods were shake-centrifugation (SC), generator column

(GCol) and slow-stir-flask (SSF) and the estimation methods were SPARC (SPARC Performs Automated Reasoning in Chemistry; Karickhoff et al., 1989) and CLOGP (Chou and Jurs, 1979). Data were also extracted from the literature. The SC method is a standard procedure in the Organization for Economic Cooperation and Development (OECD) guidelines for testing chemicals, therefore it has regulatory precedence.

TABLE 2-1. FLUORANTHENE MEASURED AND ESTIMATED $\log_{10}K_{ow}$ VALUES.

METHOD	$\log_{10}K_{ow}$	REFERENCE
Measured	5.15	De Bruijn et al., 1989
Estimated	4.90	Mabey et al., 1982
Estimated	4.95	CLOGP ^a
Estimated	4.95	GEMS ^b
Estimated	5.21	SPARC ^c
Estimated	5.22	Yalkowsky et al., 1983
Estimated	5.29	Mackay et al., 1980
Estimated	5.33	Callahan et al., 1979

^aCLOGP 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).

^bGEMS is the Graphical Exposure Modeling System for the U.S. EPA. It is an interactive computer system located on the VAX Cluster in the National Computer Center in Research Triangle Park, North Carolina, under management of EPA's Office of Toxic Substances. PC GEMS is the version for personal computers.

^cSPARC is from SPARC Performs Automated Reasoning in Chemistry, (Karickhoff et al., 1989).

In the examination of the literature data for fluoranthene, only one primary reference was found listing a measured $\log_{10}K_{ow}$. This value was 5.15 (Table 2-1). Several primary references were found for estimated $\log_{10}K_{ow}$ values in the literature ranging from 4.90 to 5.33 (Table 2-1). Although the range of reported values for fluoranthene 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 fluoranthene is 5.21. The CLOGP program estimate of the $\log_{10}K_{ow}$ value for fluoranthene using structure activity relationships is 4.95.

We had little confidence in the available measured or estimated values for K_{ow} , therefore the SC, GCol, SSF methods were used to provide additional data from which to define K_{ow} for fluoranthene (Table 2-2). The SC method yielded a $\log_{10}K_{ow} = 5.00$ ($n=3$), the GCol method yielded a $\log_{10}K_{ow} = 5.39$ ($n=4$), and the SSF method yielded a $\log_{10}K_{ow} = 5.09$ ($n=6$). Comparison of the results from the SC, GCol, SSF and SPARC K_{ow} determination methods for the five chemicals for which SQC are currently being developed (acenaphthene, dieldrin, endrin, fluoranthene and phenanthrene) indicate that the SSF method provides the best estimate of K_{ow} (U.S. EPA, 1993a). The SSF method had less variability, less experimental bias (Bias is defined as the mean difference between the best-fit estimate of K_{ow} using all four methods and the estimates from each method.) and was generally in the range of the SC, GCol, and SPARC methods (U.S. EPA, 1993a). Therefore, the SSF value of 5.09 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 six K_{ow} measurements made by SSF. The logs of the K_{ow} values measured by SSF range from 4.98 to 5.23.

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

SHAKE- CENTRIFUGATION (SC)	GENERATOR COLUMN (GCol)	SLOW-STIR FLASK (SSF)
4.99	5.19	4.98
5.00	5.35	5.02
5.01	5.47	5.02
	5.48	5.10
		5.14
		5.23
5.00 ^a	5.39 ^a	5.09 ^a

^a \log_{10} of mean of measured values.

2.3 DERIVATION OF K_{oc} FROM ADSORPTION STUDIES:

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

2.3.1 K_{oc} FROM PARTICLE SUSPENSION STUDIES:

Laboratory studies to characterize adsorption are generally conducted using particle suspensions. The high concentrations of solids and turbulent conditions necessary to keep the

mixture in suspension make data interpretation difficult as a result of a particle interaction effect. This effect suppresses the partition coefficient relative to that observed for undisturbed sediments (Di Toro, 1985; Mackay and Powers, 1987).

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 :

$$K_p = \frac{f_{oc} K_{oc}}{1 + mf_{oc} K_{oc} / v_x} \quad (2-4)$$

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:

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

Experimental data demonstrating the effect of particle suspensions was not found for fluoranthene during a comprehensive literature search for partitioning information. Nonetheless the above discussion highlights the need to consider particle interaction effects when interpreting partitioning data in particle suspension studies.

In the absence of particle effects, K_{oc} is related to K_{ow} via Equation 2-5. For $\log_{10} K_{ow} = 5.09$ (See section 2.2), this expression results in an estimate of $\log_{10} K_{oc} = 5.00$.

2.3.2 K_{oc} FROM SEDIMENT TOXICITY TESTS:

Measurements of K_{oc} are available from sediment toxicity tests using fluoranthene (Swartz et al., 1990, DeWitt et al., 1992). The tests from Swartz et al. (1990) used three

saltwater sediments with a range of organic carbon contents of 0.2 to 0.5 percent. The tests from DeWitt et al. (1992) used a marine sediment amended with five types of organic matter yielding sediments with organic carbon contents of 0.3 to 0.43 percent. (Table 4-1; Appendix B.) Fluoranthene concentrations were measured in the sediment and pore waters providing the data necessary to calculate the partition coefficient for an undisturbed bedded sediment. Swartz et al. (1990) measured total pore water concentrations and DeWitt et al. (1992) measured total and freely dissolved pore water concentrations. The relationship of total to free pore water in the DeWitt et al. (1992) analysis was applied to the total pore water data in Swartz et al. (1990) to estimate free pore water in that study. Also an adjustment was made to pore water to account for 28% loss due to filtration during experimental measurements (Swartz, 1992).

The upper panel of Figure 2-1 is a plot of the organic carbon-normalized sorption isotherm for fluoranthene, where the sediment fluoranthene concentration, ($\mu\text{g/gOC}$) is plotted versus free (dissolved) pore water concentration, ($\mu\text{g/L}$). The data used to make this plot are included in Appendix B. One of the amended sediments for the DeWitt analysis was omitted because the author believed that elevated dissolved organic carbon may have interfered with the method used to determine free interstitial water fluoranthene concentrations. The line of unity slope corresponding to the $\log_{10}K_{OC} = 5.00$ derived from SSF is compared to the data. A probability plot of the observed experimental $\log_{10}K_{OC}$ values is shown in the lower panel of Figure 2-1. The $\log_{10}K_{OC}$ values are approximately normally distributed with a mean of $\log_{10}K_{OC} = 5.05$ and a standard error of the mean of 0.071. This value agrees with the $\log_{10}K_{OC}$ of 5.00, which was computed from the SSF determined (Section 2.2) fluoranthene $\log_{10}K_{OW}$ of 5.09 using Equation 2-5.

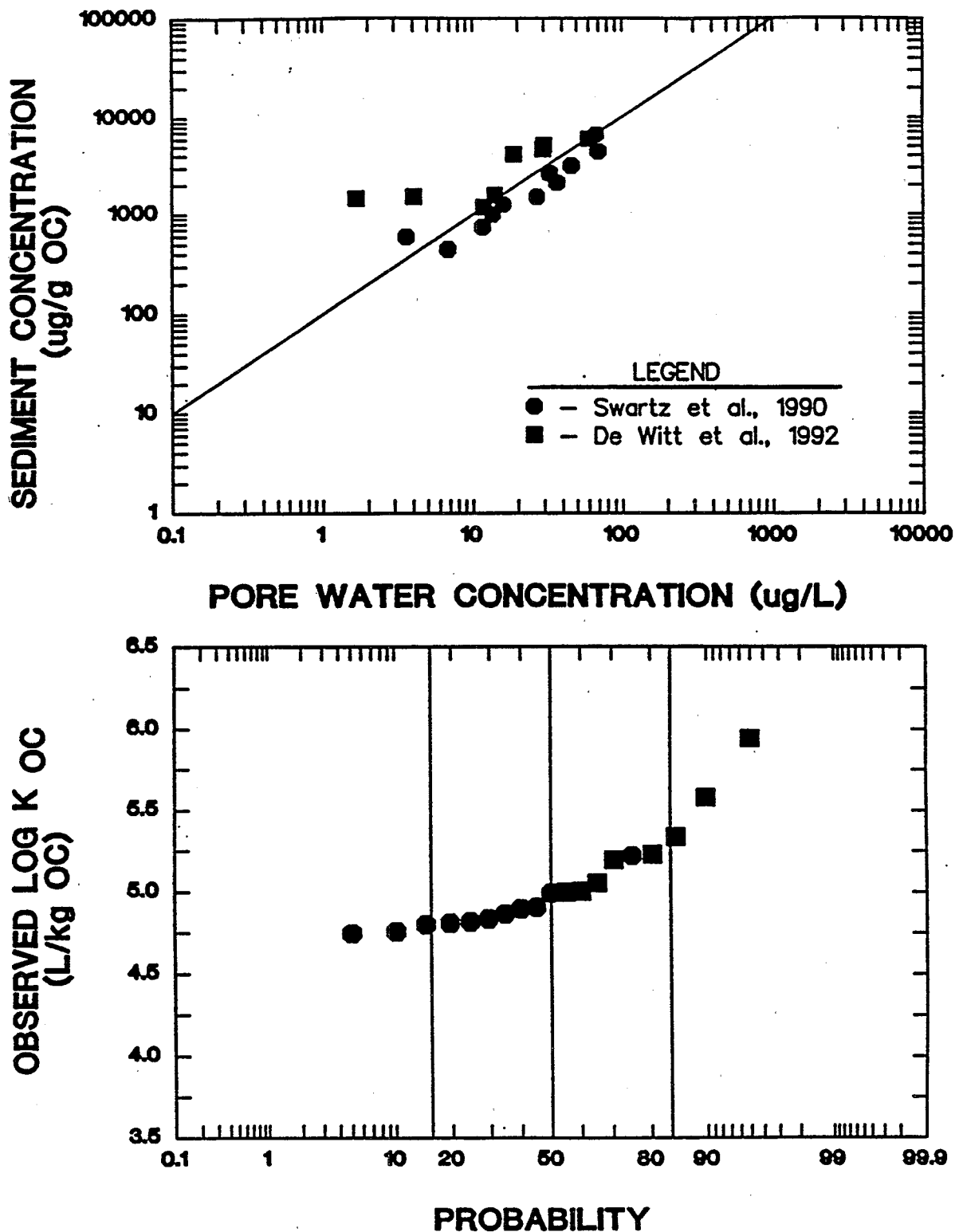


Figure 2-1. Organic carbon-normalized sorption isotherm for fluoranthene (top) and probability plot of K_{oc} (bottom) from sediment toxicity tests conducted by Swartz et al., (1990) and DeWitt et al., (1992). The line in the top panel represents the relationship predicted with a $\log K_{oc}$ of 5.00, that is $C_{s,oc} = K_{oc} \cdot C_d$.

2.4 SUMMARY OF DERIVATION OF K_{OC} FOR FLUORANTHENE:

The K_{OC} selected to calculate the SQC for fluoranthene is based on the regression of $\log_{10}K_{OC}$ to $\log_{10}K_{OW}$ (Equation 2-5), using the fluoranthene $\log_{10}K_{OW}$ of 5.09 recently measured by ERL, A. This approach, rather than the use of the K_{OC} from the toxicity tests, was adopted because the regression equation is based on the most robust data set available that spans a broad range of chemicals and particle types, thus encompassing a wide range of K_{OW} and f_{OC} . The regression equation yields a $\log_{10}K_{OC}$ of 5.00. This value is in agreement with the $\log_{10}K_{OC}$ of 5.05 measured in the sediment toxicity tests.

SECTION 3

TOXICITY OF FLUORANTHENE: WATER EXPOSURES

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

The equilibrium partitioning (EqP) method for the derivation of sediment quality criteria (SQC) uses the fluoranthene water quality criterion (WQC) final chronic value (FCV) and partition coefficients (K_{oc}) to estimate the maximum concentrations of nonionic organic chemicals in sediments, expressed on an organic carbon basis, that will not cause adverse effects to benthic organisms. For this document, life stages of species classed as benthic are either species that live in the sediment (infauna) or on the sediment surface (epibenthic) and obtain their food from either the sediment or water column (U.S. EPA, 1989c). In this section (1) the FCV from the fluoranthene 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. Tests conducted in the dark or under normal laboratory lighting are discussed in Sections 3.2 and 3.3. The phototoxicity of fluoranthene is discussed in Section 3.4.

3.2 ACUTE TOXICITY - WATER EXPOSURES:

Twenty standard acute toxicity tests with fluoranthene have been conducted under normal laboratory lighting on 13 freshwater species from 12 genera (Appendix A). Overall genus mean acute values (GMAVs) range from 36 to $> 211.7 \mu\text{g/L}$. The acute values for three species tested, Lumbriculus variegatus, Physella virgata, and Ophiogomphus sp., were all $> 178.5 \mu\text{g/L}$. The acute values for the 4 most sensitive species differed by only a factor of 3; 32 to 95

$\mu\text{g/L}$. Nine tests on benthic life-stages of nine species from eight genera are contained in this database (Figure 3-1; Appendix A). Benthic organisms were similar to water column species in sensitivity to fluoranthene; GMAVs range from 61.1 to $>178.5 \mu\text{g/L}$. One infaunal species, the annelid L. variegatus, had an LC50 of $>178.5 \mu\text{g/L}$. The final acute value (FAV) derived from the overall GMAVs (Stephan et al., 1985) for freshwater organisms is $33.58 \mu\text{g/L}$ (Table 3-2).

Fifteen acute tests have been conducted on 8 saltwater species from 8 genera (Appendix A). Overall GMAVs range from 51.54 to $>20,000 \mu\text{g/L}$, with crustaceans the most sensitive. There are results from 6 tests on benthic life-stages of 6 species from 6 genera (Figure 3-2; Appendix A). Benthic organisms were among both the most sensitive and most resistant saltwater genera to fluoranthene. The most sensitive benthic species is the mysid, Mysidopsis bahia, with an average flow-through 96 hour LC50 of $51.54 \mu\text{g/L}$ based on two tests with measured concentrations. Other benthic species for which there are data are only slightly less sensitive while others are resistant to fluoranthene and had GMAVs ranging from 66.93 to $>20,000 \mu\text{g/L}$. The FAV derived from the overall GMAVs (Stephan et al., 1985) for saltwater organisms is $16.13 \mu\text{g/L}$ (Table 3-2).

3.3 CHRONIC TOXICITY - WATER EXPOSURES:

Chronic toxicity tests have been conducted with fluoranthene using a freshwater cladoceran, (Daphnia magna) and fathead minnows (Pimephales promelas) and a saltwater mysid M. bahia (Table 3-1). D. magna and M. bahia were tested in life-cycle exposures. P. promelas were exposed in an early life-stage toxicity test.

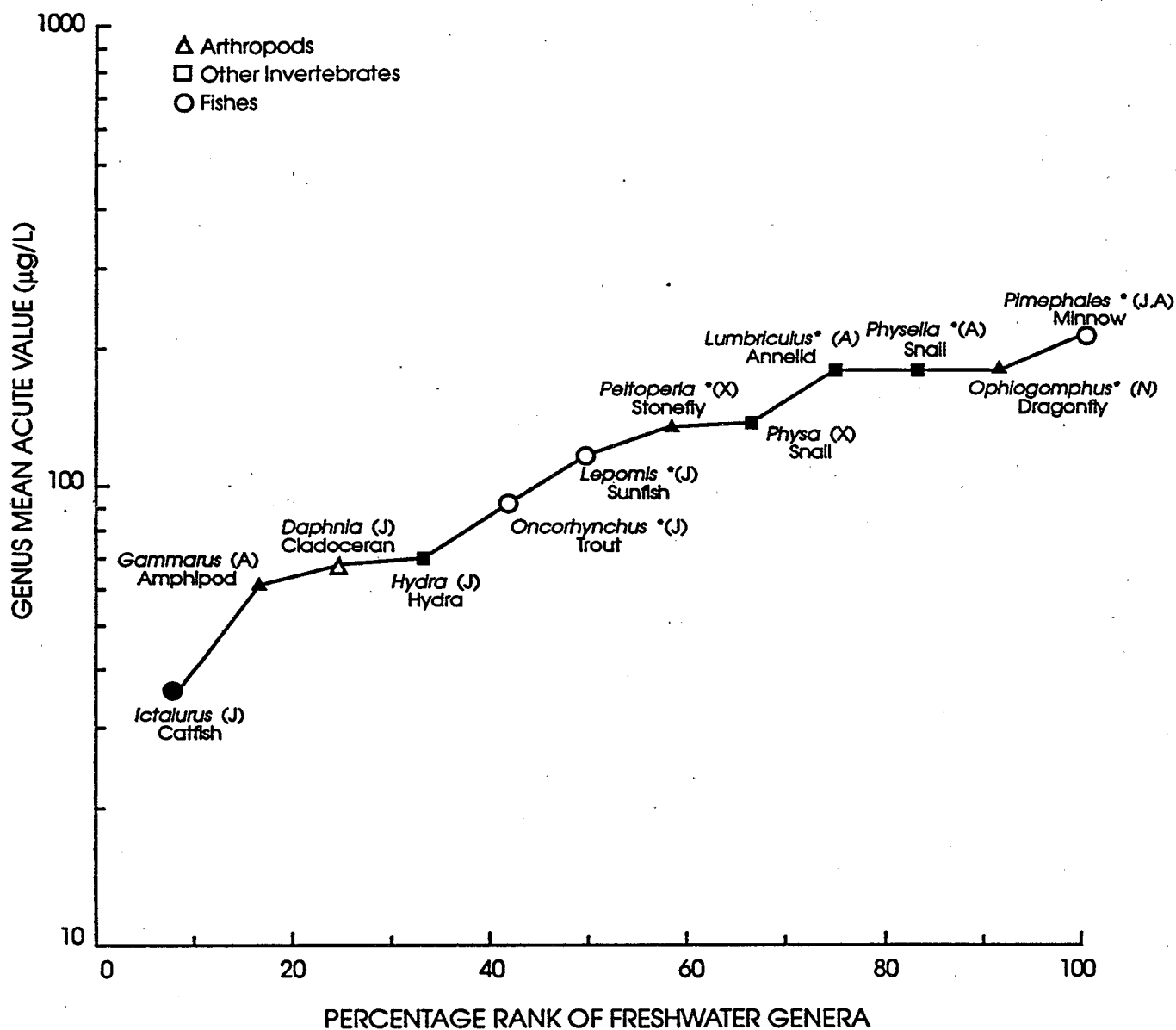


Figure 3-1. Genus mean acute values from water only acute toxicity tests using freshwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. A = adult, J = juvenile,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

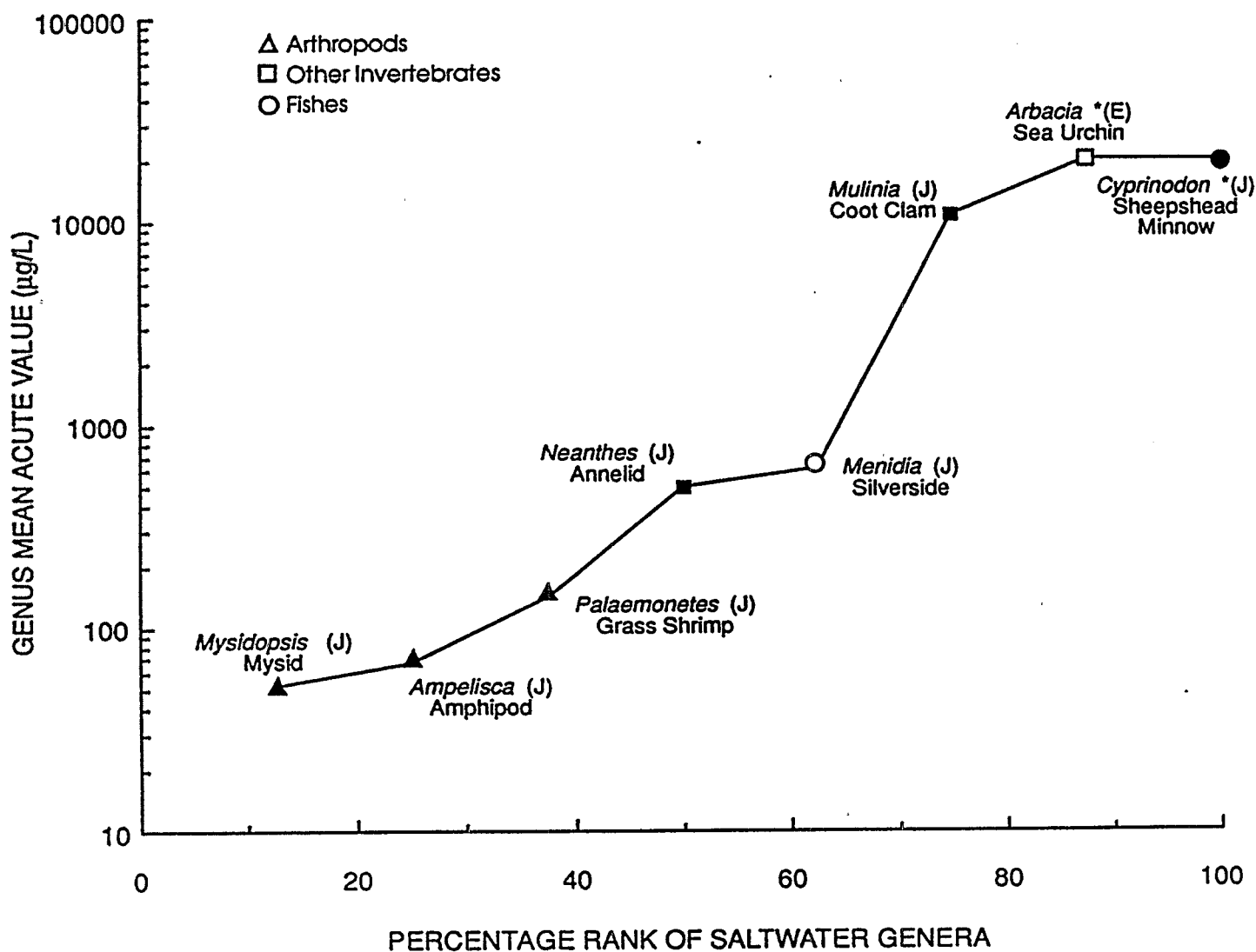


Figure 3-2. Genus mean acute values from water only acute toxicity tests using saltwater species vs. percentage rank of their sensitivity. Symbols representing benthic species are solid, those representing water column species are open. Asterisks indicate greater than values. E = embryo, J = juvenile.

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

Common Name, Scientific Name	Test* (Life stage)	Habitat* (Life stage)	NOEC*	Parental LOEC*	Parental Effect*	Progeny LOEC*	Progeny Effect*	References
			µg/L	µg/L		µg/L		
<u>FRESHWATER SPECIES: DARK</u>								
Cladoceran, <u>Daphnia magna</u>	LC	W(J,A)	<3.8 - 10.6	21.2 43.5	23.1% G 36.9% M	-	-	Brooke, 1992
Fathead minnow, <u>Pimephales promelas</u>	ELS	W(E,L,J)	3.7 - 10.4	21.7	67% M 50.2% G	-	-	Brooke, 1991
<u>SALTWATER SPECIES: DARK</u>								
Mysid, <u>Mysidopsis bahia</u>	LC	E(J,A)	<5 - 12	21 43	26.7% M 91.7% R 100% M	-	-	EG&G, 1978
Mysid, <u>Mysidopsis bahia</u>	LC	E(J,A)	0.11 - 11.1	18.8	30% M 12% G 100% R	-	-	Champlin and Poucher, 1991b
<u>FRESHWATER SPECIES: UV</u>								
Cladoceran, <u>Daphnia magna</u>	LC	W(J,A)	<0.26 - 0.88	-	-	-	-	Brooke, 1991
Fathead minnow, <u>Pimephales promelas</u>	ELS	W(E,L,J)	0.37 - 1.4	4.8	91.6% M	-	-	Brooke, 1991
<u>SALTWATER SPECIES: UV</u>								
Mysid <u>Mysidopsis bahia</u>	LC	E(J,A)	0.076	0.574	-	-	-	Champlin and Poucher, 1992b

*Test: LC = lifecycle, PLC = partial lifecycle, ELS = early life stage

Habitat: I = infauna, E = epibenthic, W = water column

Lifestage: A = adult, J = juvenile, L = larvae, E = embryo

NOEC: no observed effect concentration(s).

LOEC: lowest observed effect concentration(s).

*Effect: percentage decrease relative to controls. M = mortality G = growth R = reproduction

D. magna exposed to fluoranthene for 21 days were not affected in 10.6 $\mu\text{g/L}$ or less. There was a reduction, relative to controls, of 23.1% in growth at 21.2 $\mu\text{g/L}$ and 36.9% in survival at 43.5 $\mu\text{g/L}$. P. promelas exposed to fluoranthene for 28 days in an early life-stage toxicity test were not affected in 10.4 $\mu\text{g/L}$ or less. There was a reduction of 67% in survival and a 50.2% reduction in growth relative to controls in 21.7 $\mu\text{g/L}$. In the 96 hour acute lethality test, 25% of the P. promelas died in 211.7 $\mu\text{g/L}$, saturation for fluoranthene (Brooke, 1991). The concentration of 211.7 $\mu\text{g/L}$ was used as the acute value for P. promelas because this value approximated an LC50 and greater concentrations could not be tested in this measured flow-through test.

Saltwater M. bahia were tested in two life-cycle toxicity tests. In the first, they were exposed to fluoranthene for 28 days (EG&G, 1978). There was no effect on survival or reproduction (growth was not measured) after 28 days of exposure to fluoranthene at concentrations $\leq 12 \mu\text{g/L}$. At a fluoranthene concentration of 21 $\mu\text{g/L}$ survival and reproduction were reduced by 26.7 and 91.7%, respectively, relative to the controls. At the highest concentration of fluoranthene, 43 $\mu\text{g/L}$, all M. bahia died. In the second test, M. bahia were exposed to fluoranthene for 31 days (Champlin and Poucher, 1991b). Effect concentrations were similar to the first test. M. bahia were not affected at fluoranthene concentrations $\leq 11.1 \mu\text{g/L}$. Survival was reduced 30%, growth 12% and reproduction 100% relative to controls, in 18.8 $\mu\text{g/L}$, the highest concentration tested.

The difference between acute and chronic sensitivity to fluoranthene, in tests where UV activation did not occur, varied minimally between species (Table 3-2). Three species mean acute-chronic ratios (ACR) are available; 3.385 for D. magna, 3.404 for M. bahia and 14.09

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 FLUORANTHENE.

Common Name, Scientific Name	Acute Value (µg/L)	Chronic Value (µg/L)	Acute-Chronic Ratio	Species Mean Acute-Chronic Ratio
<u>FRESHWATER SPECIES: DARK</u>				
Cladoceran, <i>Daphnia magna</i>	102.8	30.37	3.385	3.385
Fathead minnow, <i>Pimephales promelas</i>	>211.7	15.02	14.09	14.09
<u>SALTWATER SPECIES: DARK</u>				
Mysid, <i>Mysidopsis bahia</i>	30.53	14.44	2.114	3.404
Mysid, <i>Mysidopsis bahia</i>	87	15.87	5.482	
<u>FRESHWATER SPECIES: UV</u>				
Cladoceran, <i>Daphnia magna</i>	0.97	0.92	1.05	1.05
Fathead minnow, <i>Pimephales promelas</i>	12.22	2.59	4.72	4.72
<u>SALTWATER SPECIES: UV</u>				
Mysid <i>Mysidopsis bahia</i>	1.38	0.9248	1.492	1.492

Freshwater: Dark

Final Acute Value = 33.58 µg/L
 Final Acute-Chronic Ratio = 5.455
 Final Chronic Value = 6.156 µg/L

Saltwater: Dark

Final Acute Value = 16.13 µg/L
 Final Acute-Chronic Ratio = 5.455
 Final Chronic Value = 2.957 µg/L

FRESH AND SALT WATER: UV

Final Acute Value = 0.5067
 Final Acute-Chronic Ratio = 1.948
 Final Chronic Value = 0.260

for P. promelas. The final acute chronic ratio final (ACR) is the geometric mean of these three values (5.455).

The FCVs (Table 3-2) are used as the effect concentrations for calculating the SQC for protection of freshwater and saltwater benthic species. The FCV for freshwater organisms of 6.16 $\mu\text{g/L}$ is the quotient of the FAV of 33.6 $\mu\text{g/L}$ and the final ACR of 5.45. Similarly, the FCV for saltwater organisms of 2.96 $\mu\text{g/L}$ is the quotient of the FAV of 16.1 $\mu\text{g/L}$ and the final ACR.

3.4 PHOTOTOXICITY OF FLUORANTHENE:

Under laboratory conditions many PAHs are predicted to be not acutely toxic at or below their solubility in water (Veith et al., 1983). Under ultra-violet light (UV), however, the toxicity of some PAHs can increase by several orders of magnitude. This effect has been shown to be a result of photoactivation rather than photodegradation of the parent compound to more toxic metabolites. With some PAHs, toxicity occurs by activation by UV light of chemical present on or within an organism. Bluegills (Lepomis macrochirus) exposed to anthracene in sunlight in outdoor artificial streams died; L. macrochirus in the same stream, but downstream in the shade survived. L. macrochirus exposed in the shade died within 24 hours when placed into clean water and brought into the sunlight. Likewise, D. magna were much more sensitive to anthracene in the presence of sunlight than when exposed under laboratory light, with toxicity proportional to UV intensity (Allred and Giesy, 1985). UV-A wavelengths (UVA) were implicated as responsible for most of the photoinduced toxicity.

The mechanism by which UV light activates PAHs is the same as that for electron

excitation of plant pigments during photosynthesis by visible light. This process of excitation of PAH electrons and the probable consequence of that excitation are reviewed by Newsted and Giesy (1987). Briefly, if a compound absorbs light, then electrons can be elevated to higher energy states to form the excited singlet state. If the excited electrons return immediately to their ground state then the extra energy is lost harmlessly through fluorescence. However, if the electrons pass through a triplet state, then the energy can be transferred to other molecules (thought to be oxygen in the case of PAHs). Singlet oxygen formed in this process is capable of denaturing biomolecules. Singlet oxygen is very reactive with water and unless organisms, PAH, and sunlight are present simultaneously, photoactivation does not enhance toxicity. Therefore, benthic organisms which remain buried or organisms in the shade can survive PAH concentrations which would be lethal if they emerged from the sediment or shade into sunlight. PAHs are concentrated in the non-polar environments of cells, such as the phospholipids of membranes. Singlet oxygen in tissues is longer lived, thus greatly increasing the likelihood that it would denature biomolecules. This also explains why membrane damage is one of the probable mechanisms for this type of toxicity (Kagan et al., 1987) and why organisms exposed to PAHs out of direct sunlight die when placed in the sun in PAH-free water.

Fluoranthene has exhibited photoinduced toxicity during standardized toxicity tests with a variety of organisms (Appendix A). Although, the toxicity of fluoranthene appears to increase with increases in intensity of UVA at low UV intensities (Figure 3-3), the acute toxicity of fluoranthene to saltwater organisms is similar under commercially available UV lights and sunlight (Figure 3-3; Appendix A). This is important since conducting acute and particularly chronic toxicity tests outside in sunlight would be extremely difficult and expensive. The

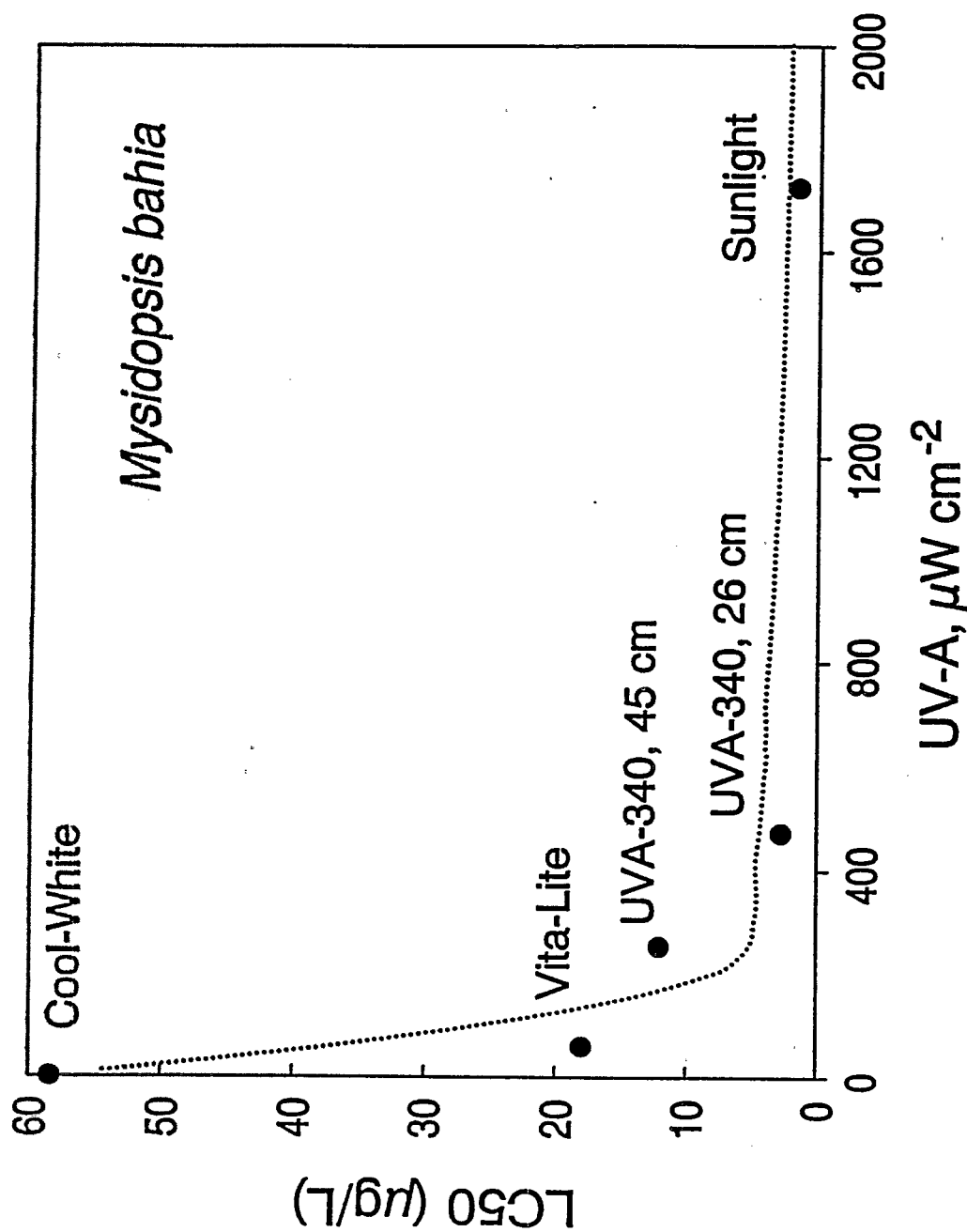


Figure 3-3. Effect of UV light on toxicity of fluoranthene to the mysid, *Mysidopsis bahia*. Labels refer to light source (Vita-light and UVA-340 are artificial UV light sources). UV measurements (in $\mu\text{W/cm}^2$) were made with a Macam Photometrics model UV-103 radiometer which read UV-A at 365 ± 36 nm. and UV-B at 310 ± 34 nm. The UV-A irradiances were 57.5 for Vita-lights at 16 cm above the water, 252 for UVA-240 at 45 cm, 473 for UVA-340 at 26 cm, and 1,725 for sunlight. The curve was similar whether the data were plotted against UV-A or UV-B. (From Thursby, 1992)

magnitude of increase in fluoranthene's toxicity following UV activation can be great. The ratio of LC50s from acute tests conducted in the dark or under cool-white fluorescent light ("dark") to LC50s for the same species exposed in the same laboratory using either UV lights or sunlight ranges from 2 to 5,000 (Appendix A). This enhanced toxicity also can occur with relatively short exposures to UV light. Kagan et al., (1985) observed that 1,000 $\mu\text{g/L}$ of fluoranthene, pyrene or anthracene was not toxic to five aquatic species in 30 minute exposures. Exposures to these PAHs for 30 minutes followed by 30 to 60 minutes exposures to UV resulted in LC50s from 4 to 360 $\mu\text{g/L}$. Data from chronic tests conducted in both the "dark" and under UV light are available for two freshwater species, D. magna and P. promelas, and one saltwater species, M. bahia. The chronic values decreased by factors of 33 for D. magna, 5.8 for P. promelas, and 2.1 for M. bahia in the presence of UV light (Table 3-2). The magnitude of increase in acute toxicity under UV light was greater than the magnitude of increase in chronic toxicity under UV light, and so the acute-chronic ratios under UV light are somewhat lower (Table 3-2).

There are not enough acute toxicity data from tests using UV light to calculate final acute and chronic values for freshwater and saltwater aquatic life. However, if existing freshwater and saltwater UV data are combined, the magnitude of the possible decrease in the FCV for organisms in photic zones can be approximated. The FAV_{UV} derived using overall GMAVs from all UV and sunlight tests (Appendix A) is 0.5067 $\mu\text{g/L}$. This value is approximately 66 times lower than the "dark" freshwater FAV, and approximately 32 times lower than the "dark" saltwater FAV.

The difference between acute and chronic sensitivity to fluoranthene, in tests under UV light, varied minimally between species (Table 3-2). Three species mean ACR are available;

1.05 for D. daphnia, 1.492 for M. bahia and 4.72 for P. promelas. The final ACR under UV light is the geometric mean of these three values (1.948). The estimated FCV_{UV} for freshwater and saltwater organisms combined of 0.2601 is the quotient of the FAV_{UV} of 0.5067 and the final ACR.

At first glance it might seem that photoinduced increases in toxicity are not relevant to benthic organisms and that SQC should not be derived using data from UV toxicity tests. This may not be true and for this reason SQC in this document may be under protective in some instances. There are many examples of specific benthic organisms where exposure to fluoranthene (and other PAHs) and sunlight can co-occur. For example, fiddler crabs which typically occupy burrows within the sediment, could accumulate fluoranthene from that sediment, and when they come out onto the surface of the sediment at low tide during daylight hours could be affected by PAHs in their tissues. Most freshwater insects that inhabit sediment during early developmental stages could also be affected by photo-induced PAHs in their tissues when they mature and emerge from water during daylight as adults. The importance of PAH's transferred from benthic species in aquatic food chains to aquatic predators which may be exposed to sunlight is unknown. Rooted aquatic plants also could be directly affected by fluoranthene contaminated sediment if they were to accumulate fluoranthene and translocate it to their leaves. Plants may be a source of photoactivated PAHs to herbivores.

At this time, U.S. EPA does not recommend a SQC value that considers fluoranthene toxicity data from UV tests. This is partly because data are insufficient to calculate a FCV. More importantly, there is an absence of data demonstrating a causal linkage between exposure of sediment-associated fluoranthene and increased risks of UV enhanced effects on benthic

organisms or organisms coupled to benthic organisms via food chains. EPA encourages research efforts on these topics.

3.5 APPLICABILITY OF THE WATER QUALITY CRITERION AS THE EFFECTS CONCENTRATION FOR DERIVATION OF THE FLUORANTHENE 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 fluoranthene-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 fluoranthene.

For fluoranthene, benthic species account for 8 out of 12 genera tested in freshwater, and 6 out of 8 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 fluoranthene was performed using the Approximate Randomization method (Noreen, 1989). The Approximate Randomization method tests the significance level of a test statistic when compared to a distribution of statistics generated from many random subsamples. The test statistic in this case is the difference between the freshwater FAV, computed from the freshwater (combined water column and benthic) species LC50 values, and the saltwater FAV, computed from the saltwater (combined water column and benthic) species LC50 values (Table 3-1). In the Approximate Randomization method, the freshwater LC50 values and the saltwater LC50 values are combined into one data set. The data set is shuffled, then separated back so that randomly generated "freshwater" and "saltwater" FAVs can be computed. The LC50 values are separated back such that the number of LC50 values used to calculate the sample FAVs are the same as the number used to calculate the original FAVs. These two FAVs are subtracted and the difference used as the sample statistic. This is done many times so that the sample statistics make up a distribution that is representative of the population of FAV differences (Figure 3-4). The test statistic is compared to this distribution to determine its 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 fluoranthene, the test-statistic falls at the 78 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

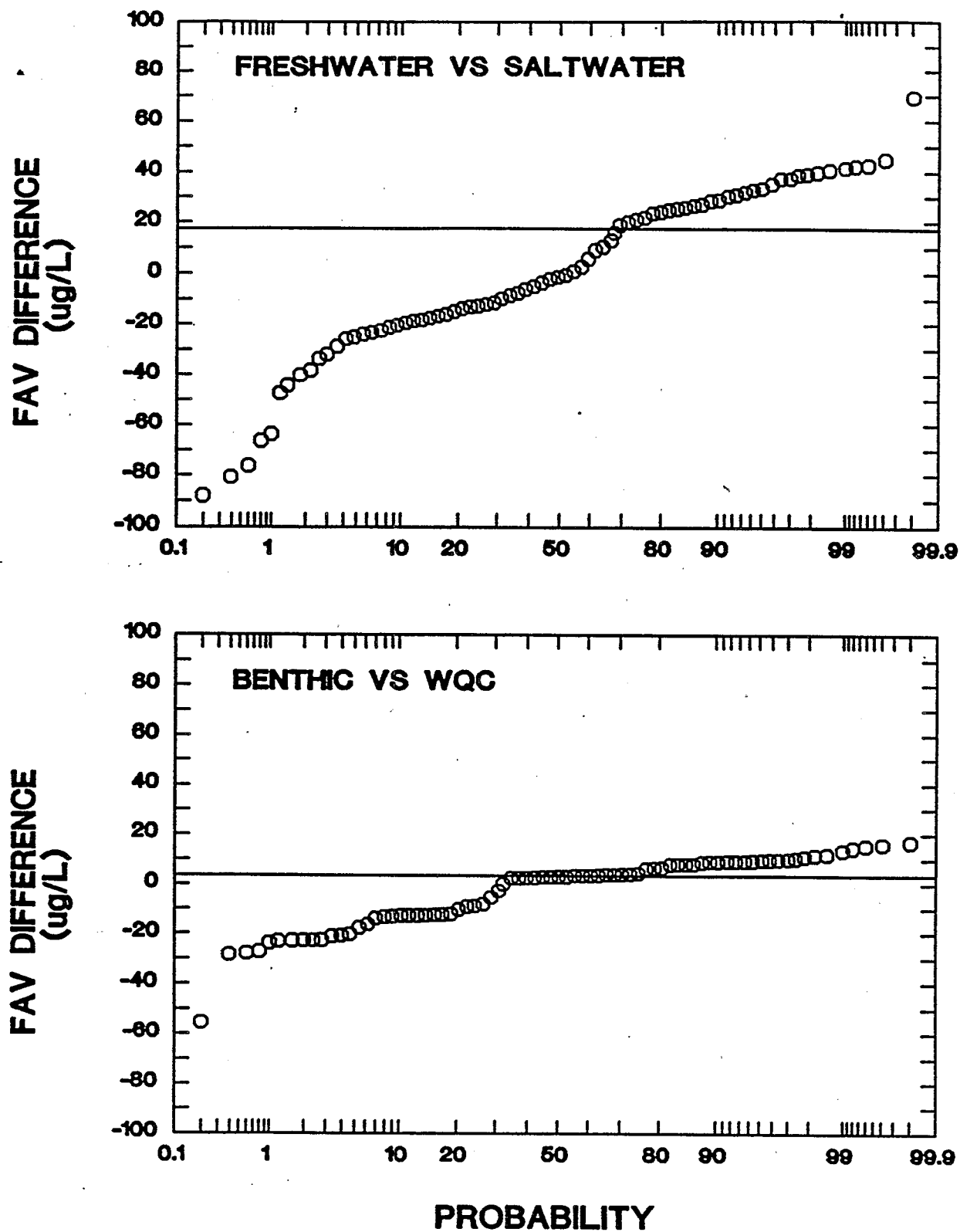


Figure 3-4. Probability distribution of FAV difference statistics to compare water-only data from freshwater vs. saltwater (upper panel) and benthic vs. WQC (lower panel) data.

TABLE 3-3. RESULTS OF APPROXIMATE RANDOMIZATION TEST FOR THE EQUALITY OF THE FRESHWATER AND SALTWATER LC50 DISTRIBUTIONS FOR FLUORANTHENE 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 (12)	Salt (8)	17.4	78
Benthic vs Water Column + Benthic (WQC)	Benthic (14)	WQC (20)	3.64	74

^aValues in parentheses are the number of LC50 values used in the comparison.

^bAR statistic = FAV difference between original compared groups.

^cProbability that the theoretical AR statistic \leq than the observed AR statistic given that the samples came from the same population.

Since freshwater and saltwater species showed similar sensitivity, a test of difference in sensitivity for benthic and all (benthic and water column species combined, hereafter referred to as "WQC") organisms combining freshwater and saltwater species using the Approximate Randomization method was performed. The test statistic in this case is the difference between the WQC FAV, computed from the WQC LC₅₀ values, and the benthic FAV, computed from the benthic organism LC₅₀ values. This is slightly different than the previous test for saltwater and freshwater species. The difference is that saltwater and freshwater species represent two separate groups. In this test the benthic organisms are a subset of the WQC organisms set. In the Approximate Randomization method for this test, the number of data points coinciding with

the number of benthic organisms are selected from the WQC data set. A "benthic" FAV is computed. The original WQC FAV and the "benthic" FAV are then used to compute the difference statistic. This is done many times and the distribution that results is representative of the population of FAV difference statistics. The test statistic is compared to this distribution to determine its level of significance. The probability distribution of the computed FAV differences are shown in the bottom panel of Figure 3-4. The test statistic for this analysis falls at the 74 percentile and the hypothesis of no difference in sensitivity is accepted (Table 3-3). This analysis suggests that the FCV for fluoranthene based on data from all tested species is an appropriate effects concentration for benthic organisms.

SECTION 4

TOXICITY OF FLUORANTHENE (ACTUAL AND PREDICTED): SEDIMENT EXPOSURES

4.1 TOXICITY OF FLUORANTHENE IN SEDIMENTS:

The toxicity of fluoranthene spiked into sediments has been tested with three saltwater amphipod species, and one amphipod, one midge, one cladoceran, and two fish species from freshwater. Data from all species tested have been included in Table 4-1, but only data from tests with benthic species have been included in Table 4-2 and Figures 4-1 and 4-2. All concentrations of fluoranthene 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 fluoranthene 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 fluoranthene. Data from these studies have not been included here because it is not possible to determine the contribution of fluoranthene to the observed toxicity.

Gendusa (1990) exposed fathead minnows, Pimephales promelas, and channel catfish, Ictalurus punctatus, to fluoranthene-spiked sediments with a total organic carbon content (TOC)

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

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method/ Duration (Days)	Response	Sediment Fluoranthene		Pore Water LC50, µg/L	References
					Dry wt.	LC50, µg/g Org. Car.		
FRESHWATER								
Amphipod, <u>Hyalolella azteca</u>	Water Research Field Station Denton Co., Texas	0.46	S, M/10	EC50	2.3	500	45.9	Suedel et al., 1993; Suedel, 1989
Amphipod, <u>Hyalolella azteca</u>	Lake Fork Reservoir Wood Co., Texas	0.50	S, M/10	EC50	7.4	1,480	236.5	Suedel et al., 1993; Suedel, 1989
Amphipod, <u>Hyalolella azteca</u>	Trinity River Navarro Co., Texas	0.44	S, M/10	EC50	5.5	1,250	97.6	Suedel et al., 1993; Suedel, 1989
Midge, <u>Chironomus tentans</u>	Water Research Field Station Denton Co., Texas	0.46	S, M/10	EC50	7.3	1,587	91.2	Suedel et al., 1993; Suedel, 1989
Midge, <u>Chironomus tentans</u>	Lake Fork Reservoir Wood Co., Texas	0.50	S, M/10	EC50	8.7	1,748	251.0	Suedel et al., 1993; Suedel, 1989
Midge, <u>Chironomus tentans</u>	Trinity River Reservoir Wood Co., Texas	0.44	S, M/10	EC50	3.0	682	75.7	Suedel et al., 1993; Suedel, 1989
Cladoceran, <u>Daphnia magna</u>	Water Research Field Station Denton Co., Texas	0.46	S, M/10	EC50	15.0	3,261	158.0	Suedel et al., 1993; Suedel, 1989
Cladoceran, <u>Daphnia magna</u>	Lake Fork Reservoir Wood Co., Texas	0.50	S, M/10	EC50	11.9	2,380	197.3	Suedel et al., 1993; Suedel, 1989
Cladoceran, <u>Daphnia magna</u>	Trinity River Reservoir Wood Co., Texas	0.44	S, M/10	EC50	4.2	955	88.6	Suedel et al., 1993; Suedel, 1989
Fathead Minnow, <u>Pimephales promelas</u>	Water Research Field Station, University of North Texas	0.70	S, M/4	LC50	0.97	131.0	-	Gendusa, 1990
Fathead Minnow, <u>Pimephales promelas</u>	Water Research Field Station, University of North Texas	0.70	S, M/30	LC50	0.437	62.46	-	Gendusa, 1990

TABLE 4-1: (continued)

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method*/ Duration (Days)	Response	Sediment Fluoranthene LC50, µg/g Dry wt. Org. Car.	Pore Water LC50, µg/L	References
Channel Catfish, <i>Ictalurus punctatus</i>	Water Research Field Station, University of North Texas	0.70	S, M/4	LC50	3.683	526.2	Gendusa, 1990
Amphipod, <i>Corophium spinicorne</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.18	S, M/10	LC50	5.1	2,830	Swartz et al., 1990
Amphipod, <i>Corophium spinicorne</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.31	S, M/10	LC50	>13.6	>4,390	Swartz et al., 1990
Amphipod, <i>Corophium spinicorne</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.48	S, M/10	LC50	>13.6	>2,830	Swartz et al., 1990
Amphipod, <i>Rhepoxynius abronius</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.18	S, M/10	LC50	3.4	1,890	Swartz et al., 1990
Amphipod, <i>Rhepoxynius abronius</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.31	S, M/10	LC50	6.5	2,100	Swartz et al., 1990
Amphipod, <i>Rhepoxynius abronius</i>	Yaquina Bay, OR, Sand with Organic Matter Added	0.48	S, M/10	LC50	10.7	2,230	Swartz et al., 1990
Amphipod, <i>Hyaella azteca</i>	Fine Sand, 0.5mm Sieved (2ppt Salinity)	-	S, M/10	LC50	15.4 (21.2) ^b	-	DeWitt et al., 1989
Amphipod, <i>Eohaustorius estuarius</i>	Fine Sand, 0.5mm Sieved (2ppt Salinity)	-	S, M/10	LC50	9.3 (13.8) ^b	-	DeWitt et al., 1989
Amphipod, <i>Eohaustorius estuarius</i>	Fine Sand, 0.5mm Sieved (5ppt Salinity)	-	S, U/10	LC50	14.0 ^b	-	DeWitt et al., 1989
Amphipod, <i>Eohaustorius estuarius</i>	Fine Sand, 0.5mm Sieved (10ppt Salinity)	-	S, M/10	LC50	10.7 (15.1) ^b	-	DeWitt et al., 1989

TABLE 4-1: (continued)

Common/Sci. Name	Sediment Source; Description	TOC (%)	Method/ Duration (Days)	Response	Sediment Fluoranthene LC50, µg/g Dry wt. Org. Car.	Pore Water LC50, µg/L	References
Amphipod, <i>Rehastorius estuarius</i>	Fine Sand, 0.5mm Sieved (15ppt Salinity)	-	S,U/10	LC50	13.9 ^b	-	DeWitt et al., 1989
Amphipod, <i>Rehastorius estuarius</i>	Fine Sand, 0.5mm Sieved (28ppt Salinity)	-	S,M/10	LC50	11.8 (17.5) ^b	-	DeWitt et al., 1989
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved (28ppt Salinity)	-	S,M/10	LC50	5.1 (6.6) ^b	-	DeWitt et al., 1989
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved, <i>Zostera</i> added	0.34	S,U/10	LC50	19.1 ^b (>14.8) ^a	>315 ^d (>179) ^a	DeWitt et al., 1992
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved, "Suspended Solids" Added	0.34	S,M/10	LC50	15.0	4,410	DeWitt et al., 1992
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved, Mud Added	0.40	S,M/10	LC50	12.6	3,150	DeWitt et al., 1992
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved, Oyster Feces Added	0.31	S,M/10	LC50	9.56	3,080	DeWitt et al., 1992
Amphipod, <i>Rhepoxynius abronius</i>	Fine Sand, 0.5mm Sieved, Shrimp Feces Added	0.31	S,M/10	LC50	8.65	2,790	DeWitt et al., 1992

^aS = Static, M = Measured concentration, U = Unmeasured.^bNominal LC50 value^cMeasured LC50 value^dTotal interstitial water^eFree interstitial water

of 0.7%. Sediments were spiked with fluoranthene dissolved in acetone which was added directly to the sediment. After the acetone was allowed to evaporate for 15 minutes, crystalline fluoranthene was stirred into the sediment, and overlying water was added. Animals were added 24 hours later. I. punctatus were exposed for 96 hours, P. promelas were exposed for 96 hours and 30 days. Fluoranthene concentrations at the end of the sediment tests were much lower than at the beginning of the tests, presumably because the sediments did not have time to equilibrate. The 4-day LC50s for I. punctatus and P. promelas and the 30-day LC50 for P. promelas were 3.683, 0.97 and 0.437 $\mu\text{g/g}$ dry wt. respectively, and 526.2, 131.0, and 62.46 $\mu\text{g/g}$ organic carbon respectively. These LC50s are means of the LC50s calculated on the basis of the initial and final measurements of fluoranthene in the tests.

Suedel et al., (1993) exposed the amphipod, Hyalella azteca, the midge, Chironomus tentans, and the cladoceran, Daphnia magna, to three fluoranthene-spiked sediments of similar TOC content, 0.44 to 0.50%. Sediments were spiked with fluoranthene dissolved in acetone. Microliter quantities were added to the sediment surface and allowed to dry for two minutes, then the sediments were mixed by hand for 60 seconds before the addition of test water. Animals were added to the test beakers 18-24 hours later. LC50s normalized to dry wt. differed by a factor of 3.2 (2.3 to 7.4 $\mu\text{g/g}$) for H. azteca, a factor of 2.9 (3.0 to 8.7 $\mu\text{g/g}$) for C. tentans, and a factor of 3.6 (4.2 to 15.0 $\mu\text{g/g}$) for D. magna. The organic carbon normalized LC50s for H. azteca differed by a factor of 3.0 (500 to 1,480 $\mu\text{g/g}_{\text{oc}}$), a factor of 2.6 (682 to 1,740 $\mu\text{g/g}_{\text{oc}}$) for C. tentans, and a factor of 3.4 (955 to 3,261 $\mu\text{g/g}_{\text{oc}}$) for D. magna. Organic carbon normalization had little effect because the TOC contents of the three sediments were so similar.

Swartz et al. (1990) exposed the amphipods Corophium spinicorne and Rhepoxynius abronius to three fluoranthene-spiked sediments with TOC contents of 0.18%, 0.31% and 0.48%. Sediments were prepared using the methods of Swartz et al. (1985) by mixing varying amounts of organically-rich fine sediment into sand with a low organic content. Fluoranthene, dissolved in acetone, was added to sediment aliquots in rolling mill jars and rolled. The sediments were allowed to equilibrate for approximately two weeks before sediment was added to test chambers. The 10-day LC50's for R. abronius increased with increasing organic carbon concentration when the fluoranthene concentration was expressed on a dry weight basis, but were not different when concentration was expressed on an organic carbon basis. LC50's normalized to dry weight differed by a factor of 3.1 (3.4 to 10.7 $\mu\text{g/g}$) for R. abronius over a 2.7-fold range of TOC. The organic carbon normalized LC50's for R. abronius differed by a factor of 1.2 (1,890 to 2,230 $\mu\text{g/g}_{\text{OC}}$). Because less than 50% mortality of C. spinicorne resulted in the highest fluoranthene treatments in two of the three sediments used in this experiment, it was not possible to make similar comparisons with this species.

De Witt et al. (1989) exposed the saltwater amphipod E. estuarius to fluoranthene-spiked sediments at five different salinities and R. abronius and the freshwater amphipod H. azteca to fluoranthene-spiked sediments at single salinities (Table 4-1). Sediments were spiked with fluoranthene dissolved in acetone and mixed on a rolling mill intermittently over a 24 hour period. Overlying water was then added to the test chambers and allowed to equilibrate for 24 hours before the addition of test animals. Fluoranthene toxicity to E. estuarius was not affected by interstitial water salinity. Nominal LC50 values (fluoranthene was not measured at all salinities) varied by a factor of 1.3 (range 13.8 to 17.5 $\mu\text{g/g}$) on a dry weight basis. TOC was

not measured in these sediments. The 10-day LC50 for *R. abronius* (5.1 $\mu\text{g/g}$, dry wt) was similar to those reported by Swartz et al. (1990).

De Witt et al. (1992) exposed *R. abronius* to five fluoranthene-spiked sediments of similar organic carbon content amended with organic carbon from five sources: *Zostera marina* (eelgrass); fine grained material which had settled from the water column of Yaquina Bay, OR; organic-rich sediment from a small slough in Alsea Bay, OR; feces of a suspension-feeding oyster (*Crassostrea gigas*) and feces of a deposit-feeding shrimp (*Callinassa californiensis*). Sediments were spiked by shell coating fluoranthene onto glass jars and rolling for 24 hours. The sediments were then allowed to equilibrate for 5 weeks before use in the experiments. The TOC content of the sediment varied from 3.1 to 4.0%. The authors concluded that the source of organic carbon had little effect on the 10-day LC50 values which varied by a factor of 2.2 (range 8.65 to 19.1 $\mu\text{g/g}$) on a dry weight basis and by a factor of 2.0 (range 2,790 to 5,620 $\mu\text{g/g}$ on an organic carbon basis.

Combining the results of Swartz et al., (1990) and De Witt et al. (1992) for *R. abronius*, 10-day LC50 values for fluoranthene varied by a factor of 5.6 (3.4 to 19.1 $\mu\text{g/L}$) on a dry weight basis and by a factor of 3.0 (1,890 to 5,620) on an organic carbon normalized basis (Table 4-1).

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 fluoranthene is also 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 fluoranthene and other nonionic organic chemicals on an organic carbon basis for a range of sediments. Evidence supporting this prediction for fluoranthene 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 three benthic species (Table 4-2). Data from tests with water column species were not considered in this analysis. Suedel et al., (1993) found 10-day LC50 values based on pore-water concentrations varied by a factor of 5.15 (45.9-236.5) for H. azteca and a factor of 3.32 (75.7-251.0) for C. tentans. This variability is greater than observed when using either dry weight (factors of 3.2 for H. azteca and 1.7 for C. tentans) or organic carbon (factors of 2.9 and 1.6) normalization are used. The short equilibration time used may explain these results. Swartz et al. (1990) found 10-day LC50 values based on pore-water concentrations varied by a factor of 1.3 (22.7 to 29.4 $\mu\text{g/L}$) for R. abronius. This variability is less than that shown when dry weight (factor of 3.1) normalization is used to determine LC50s based on fluoranthene concentration in sediments, but similar to that shown when organic carbon (factor of 1.2) normalization is used.

De Witt et al. (1992) found that 10-day LC50 values based on pore water concentrations varied by a factor of 2.8 (range 9.38 to 26.6 $\mu\text{g/L}$) for four of the five sediments tested, but that the pore water LC50 for the fifth sediment was much higher than that of the other four

Table 4-2: Water-only and sediment LC50s used to test the applicability of the equilibrium partitioning theory for fluoranthene.

Common/Sci. Name	Method/ Duration(days)	Water Only LC50	Overlying Water LC50	Pore Water LC50	TOC (%)	Sediment Fluoranthene			Predicted ^b LC50 µg/g OC	Ratio:			
						µg/L	µg/L	µg/L		Actual LC50	Predicted Reference		
												µg/g Dry Wt.	µg/g OC
FRESHWATER													
Amphipod, <i>Hyallela azteca</i>	S,M/10	44.9	44.7	45.9	0.46	2.3	500	4,490	0.111	Suedel et al., 1993			
Amphipod, <i>Hyallela azteca</i>	S,M/10	44.9	54.0	236.5	0.50	7.4	1,480	4,490	0.330	Suedel et al., 1993			
Amphipod, <i>Hyallela azteca</i>	S,M/10	44.9	32.4	97.6	0.44	5.5	1,250	4,490	0.278	Suedel et al., 1993			
Midge, <i>Chironomus tentans</i>	S,M/10	31.9	61.0	91.2	0.46	7.3	1,587	3,190	0.497	Suedel et al., 1993			
Midge, <i>Chironomus tentans</i>	S,M/10	31.9	50.6	251.0	0.50	8.7	1,740	3,190	0.545	Suedel et al., 1993			
Midge, <i>Chironomus tentans</i>	S,M/10	31.9	30.4	75.7	0.44	3.0	682	3,190	0.214	Suedel et al., 1993			
Channel Catfish <i>Ictalurus punctatus</i>	S,M/4	36.0 ^a	18.1	-	0.70	3.683 ^a	526.2 ^a	3,600	0.146	Gendusa, 1990			
SALTWATER													
Amphipod, <i>Rhepoxygnus abronius</i>	S,M/10	13.9 ^d	-	22.7	0.18	3.4	1,890	1,390	1.36	Swartz et al., 1990			
Amphipod, <i>Rhepoxygnus abronius</i>	S,M/10	13.9 ^d	-	29.4	0.31	6.5	2,100	1,390	1.51	Swartz et al., 1990			
Amphipod, <i>Rhepoxygnus abronius</i>	S,M/10	13.9 ^d	-	24.2	0.48	10.7	2,230	1,390	1.60	Swartz et al., 1990			
Amphipod, <i>Rhepoxygnus abronius</i>	S,M/10	13.9 ^d	-	>315 ^a	0.34	19.1 ^f	5,620 ^a	1,390	4.04	DeWitt et al., 1992			

TABLE 4-2: (continued)

Common/Sci. Name	Method/ Duration (days)	Water Only LC50	Overlying Water LC50	Pore Water LC50	TOC (%)	Sediment Fluoranthene			Predicted ^b LC50 µg/g OC	Ratio: Actual LC50 Predicted	Reference
						µg/g Dry Wt.	LC50 µg/g OC	µg/g OC			
		µg/L	µg/L	µg/L							
Amphipod, <u>Rhepoxynius abronius</u>	S,M/10	13.9 ^d	-	14.1 ^e	0.34	15.0	4,410	1,390	3.17		DeWitt et al., 1992
Amphipod, <u>Rhepoxynius abronius</u>	S,M/10	13.9 ^d	-	19.2 ^e	0.40	12.6	3,150	1,390	2.27		DeWitt et al., 1992
Amphipod, <u>Rhepoxynius abronius</u>	S,M/10	13.9 ^d	-	26.6 ^e	0.31	9.56	3,080	1,390	2.22		DeWitt et al., 1992
Amphipod, <u>Rhepoxynius abronius</u>	S,M/10	13.9 ^d	-	9.38 ^e	0.31	8.65	2,790	1,390	2.01		DeWitt et al., 1992

^aS = static, M = measured concentration^bPredicted LC50 (µg/g_w) = Water-only LC50 (µg/L) x K_{ow} (L/Kg_w) x 1Kg_w/1000g_w; where K_{ow} = 10^{3.00}^cMean of initial and final LC50^dFrom Swartz, 1991b.^eTotal interstitial water^fUnmeasured, measured value = 14.81^gUnmeasured, measured value = 4,360

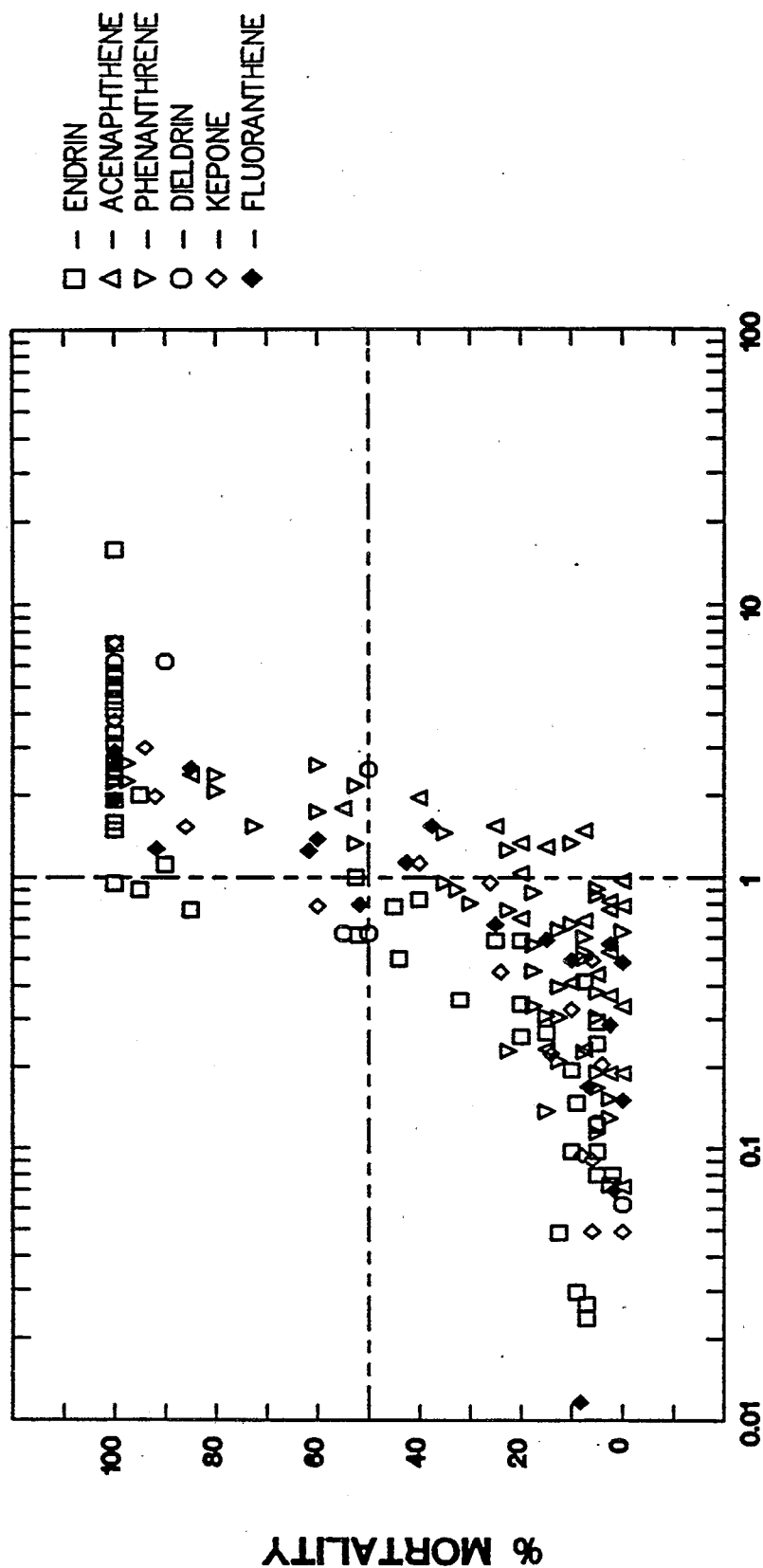
sediments ($> 315 \mu\text{g/L}$). This result runs counter to previous observations that the pore water chemical concentration shows strong correlation with toxicity. De Witt et al. (1992) note that this eelgrass amended sediment was the only sediment tested where the organic carbon originated from fresh plant material. Similarity of LC50s on an organic carbon basis and dry weight basis suggest that the pore water fluoranthene may not have been entirely bioavailable. Free (not associated with dissolved organic carbon, DOC) pore water fluoranthene concentration was measured using a modification of the Landrum et al. (1984) reverse-phase separation method. Free pore water fluoranthene concentrations were generally 60 to 90% of the total pore water concentrations. For the five sediments tested, LC50 values based on free fluoranthene in pore water ($\text{LC50} = 8.09$ to $> 179 \mu\text{g/L}$) were as variable as those based on total pore water fluoranthene (9.38 to $> 315 \mu\text{g/L}$). The fact that the organic-carbon normalized LC50s may be better predictors of toxicity than pore water concentrations was also observed with dieldrin (Hoke and Ankley, 1991). Partitioning to dissolved organic carbon was proposed to explain the lack of similarity of LC50 values based on total pore water dieldrin concentrations. This explanation is not applicable to results with fluoranthene because the total and free pore water fluoranthene concentrations and LC50 values were similar and uniform across all sediment types.

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, 1991a), endrin (Nebeker et al., 1989; Schuytema et al., 1989), dieldrin (Hoke 1992), fluoranthene (Swartz et al., 1990, DeWitt et al. 1992), or kepone (Adams et al.,

1985) (Figure 4-1; Appendix B). The data included in this analysis come from tests conducted at EPA laboratories or from tests which utilized designs at least as rigorous as those conducted at the EPA laboratories. Tests with acenaphthene and phenanthrene used two saltwater amphipods (L. plumulosus and E. estuarius) and marine sediments. Tests with fluoranthene used a saltwater amphipod (R. abronius) and marine sediments. Freshwater sediments spiked with endrin were tested using the amphipod H. azteca; while kepone-spiked sediments were tested using the midge, C. tentans. Figure 4-1 presents the percentage mortalities of the benthic species tested in individual treatments for each chemical versus "pore water toxic units" (PWTU) for all sediments tested. PWTUs are the concentration of the chemical in pore water ($\mu\text{g/L}$) divided by the water only LC50 ($\mu\text{g/L}$). Theoretically, 50% mortality should occur at one interstitial water toxic unit. At concentrations below one PWTU there should be less than 50% mortality, and at concentrations above one PWTU there should be greater than 50% mortality. Figure 4-1 shows that at concentrations below one PWTU mortality was generally low, and increased sharply at approximately one PWTU. Therefore, this comparison supports the concept that interstitial water concentrations can be used to predict the response of an organism to a chemical that is not sediment specific. This pore water normalization was not used to derive SQC in this document because of the complexation of nonionic organic chemicals with pore water DOC (Section 2) and the difficulties of adequately sampling pore waters.

4.3 TESTS OF THE EQUILIBRIUM PARTITIONING PREDICTION OF SEDIMENT TOXICITY:

SQC derived using the EqP approach utilize partition coefficients and FCVs from WQC documents to derive the SQC concentration for protection of benthic organisms. The partition



PORE WATER TOXIC UNITS

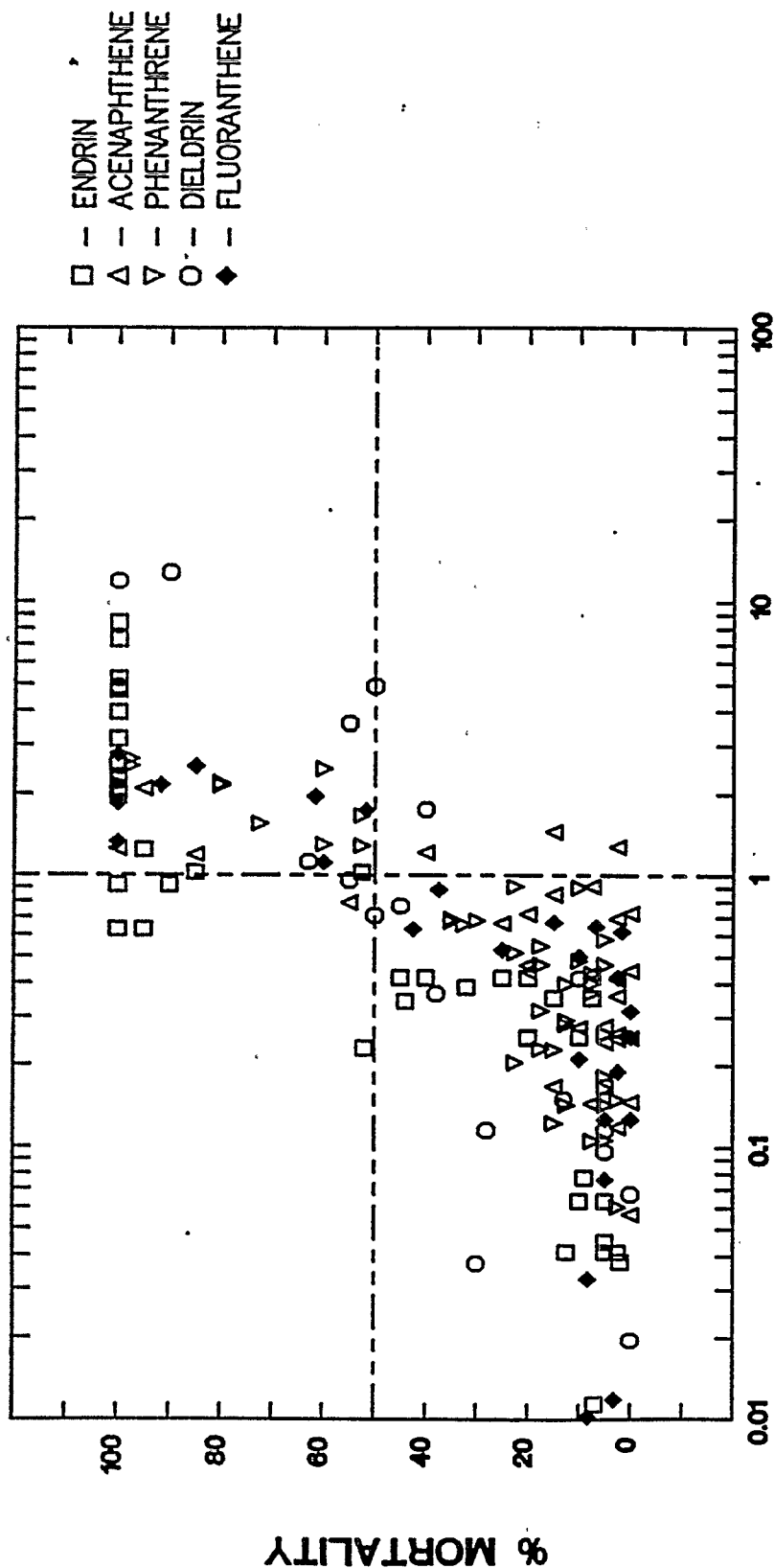
Figure 4-1. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991a), endrin (Nebeker et al., 1989; Schuytema et al., 1989), or fluoranthene (Swartz et al., 1990; De Witt et al., 1992), and midge in sediments spiked with dieldrin (Hoke, 1992) or kepone (Adams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

coefficient (K_{oc}) is used to normalize sediment concentrations and predict biologically available concentrations across sediment types. The data required to test the organic carbon normalization for fluoranthene 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 $\mu\text{g/L}$, (2) an identical sediment effect concentration on an organic carbon basis, such as a 10-day LC50 value in $\mu\text{g/g}_{oc}$, and (3) a partition coefficient for the chemical, K_{oc} in L/kg_{oc} . This section presents evidence that the observed effect concentration in sediments (2) can be predicted utilizing the water only effect concentration (1) and the partition coefficient (3).

Predicted ten-day LC50 values from fluoranthene-spiked sediments tests with H. azteca and C. tentans were calculated (Table 4-2) using the $\text{Log}_{10}K_{oc}$ value of 5.00 from Section 2 of this document and the sediment LC50's in Suedel (1993). Ratios of actual to predicted LC50s for fluoranthene averaged 0.217 (range 0.111 to 0.330) for H. azteca and 0.387 (range 0.214 to 0.545) for C. tentans. The ratio of actual to predicted LC50 for I. punctatus (Gendusa, 1990) was 0.146. Data on P. promelas (Gendusa, 1990) and D. magna were not used for prediction because they are not benthic organisms.

Predicted ten-day LC50 values on a $\mu\text{g/g}_{oc}$ basis from fluoranthene-spiked sediment tests with R. abronius (Swartz et al., 1990; De Witt et al., 1992) were calculated (Table 4-2) using the value of K_{oc} ($10^{5.00}$) from Section 2 of this document and the 10-day water-only EC50 values in Swartz (1991a). Ratios of actual to predicted LC50s for fluoranthene averaged 2.13 (range 1.36 to 4.04) for R. abronius. The data from De Witt et al (1989) can not be used for prediction because the TOC of the sediments was not measured.

A more detailed evaluation of the accuracy and precision of the EqP prediction of the response of benthic organisms can be made using the results of toxicity tests with amphipods exposed to sediments spiked with acenaphthene, phenanthrene, dieldrin, endrin, or fluoranthene. The data included in this analysis came from tests conducted at EPA laboratories or from tests which utilized designs at least as rigorous as those conducted at the EPA laboratories. Data from the kepone experiments are not included because a measured K_{ow} for kepone obtained using the slow stir flask method is not available. Swartz (1991a) 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 two freshwater dieldrin-spiked sediments having 2.0 and 1.5% organic carbon. Nebeker et al. (1989) and Schuytema et al. (1989) exposed H. azteca to three endrin-spiked sediments having 3.0, 6.1 and 11.2% organic carbon. Figure 4-2 presents the percentage mortalities of amphipods in individual treatments of each chemical versus "predicted sediment toxic units" (PSTU) for each sediment treatment. PSTUs are the concentration of the chemical in sediments ($\mu\text{g/g}_{oc}$) divided by the predicted LC50 ($\mu\text{g/g}_{oc}$) 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



PREDICTED SEDIMENT TOXIC UNIT

Figure 4-2. Percent mortality of amphipods in sediments spiked with acenaphthene or phenanthrene (Swartz, 1991a), 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 ($\mu\text{g/goc}$) divided by the predicted LC50 ($\mu\text{g/goc}$) in sediments ($K_{oc} \times \text{Water-only LC50}$ ($\mu\text{g/L}$) $\times 1 \text{ Kgoc}/1,000\text{goc}$). (See Appendix B in this document and Appendix B in the dieldrin, endrin, acenaphthene, and phenanthrene SQC documents for raw data).

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.

SECTION 5

CRITERIA DERIVATION FOR FLUORANTHENE

5.1 CRITERIA DERIVATION:

The water quality criteria (WQC) Final Chronic Value (FCV), without an averaging period or return frequency (See section 3), is used to calculate the sediment quality criteria (SQC) because it is probable that the concentration of contaminants in sediments are relatively stable over time, thus exposure to sedentary benthic species should be chronic and relatively constant. This is in contrast to the situation in the water column, where a rapid change in exposure and exposures of limited durations can occur due to fluctuations in effluent concentrations, dilutions in receiving waters or the free-swimming or planktonic nature of water column organisms. For some particular uses of the SQC it may be appropriate to use the areal extent and vertical stratification of contamination of a sediment at a site in much the same way that averaging periods or mixing zones are used in 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 fluoranthene 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 ($\mu\text{g/L}$) is the chronic concentration from the WQC for the chemical of interest, then the SQC ($\mu\text{g/g}$ sediment), is computed using the partition coefficient, K_p (L/g sediment), between sediment and pore water:

$$\text{SQC} = K_p \text{FCV} \quad (5-1)$$

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

$$\text{SQC}_{oc} = K_{oc} \text{FCV} \quad (5-2)$$

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 fluoranthene SQC.

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

TABLE 5-1. SEDIMENT QUALITY CRITERIA FOR FLUORANTHENE

Type of Water Body	Log ₁₀ K _{ow} (L/kg)	Log ₁₀ K _{oc} (L/kg)	FCV (μg/L)	SQC _{oc} (μg/g _{oc})
Fresh Water	5.09	5.00	6.16	620 ^a
Salt Water	5.09	5.00	2.96	300 ^b

$$^a\text{SQC}_{oc} = (10^{5.00} \text{ L/kg}_{oc}) \cdot (10^{-3} \text{ kg}_{oc}/\text{g}_{oc}) \cdot (6.16 \text{ } \mu\text{g fluoranthene/L}) = 620 \text{ } \mu\text{g fluoranthene/g}_{oc}$$

$$^b\text{SQC}_{oc} = (10^{5.00} \text{ L/kg}_{oc}) \cdot (10^{-3} \text{ kg}_{oc}/\text{g}_{oc}) \cdot (2.96 \text{ } \mu\text{g fluoranthene/L}) = 300 \text{ } \mu\text{g fluoranthene/g}_{oc}$$

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

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

For example, a freshwater sediment with a concentration of 6.00 μg chemical/g_{DRY WT} and 0.5 % TOC has an organic carbon-normalized concentration of 1,200 μg/g_{oc} (6.00 μg/g_{DRY WT}

• $100 \div 0.5 = 1,200 \mu\text{g/g}_{\text{OC}}$ which exceeds the SQC of $620 \mu\text{g/g}_{\text{OC}}$. Another freshwater sediment with the same concentration of fluoranthene ($6.00 \mu\text{g/g}_{\text{DRY WT}}$) but a TOC concentration of 5.0% would have an organic carbon normalized concentration of $120 \mu\text{g/g}_{\text{OC}}$ ($6.00 \mu\text{g/g}_{\text{DRY WT}} \cdot 100 \div 5.0 = 120 \mu\text{g/g}_{\text{OC}}$), which is below the SQC for fluoranthene.

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 ($\text{SQC}_{\text{DRY WT}}$). This "conversion" must be done for each level of TOC of interest:

$$\text{SQC}_{\text{DRY WT}} = \text{SQC}_{\text{OC}} (\mu\text{g/g}_{\text{OC}}) \cdot (\% \text{ TOC} \div 100)$$

where $\text{SQC}_{\text{DRY WT}}$ is the dry weight normalized SQC value. For example, the SQC value for freshwater sediments with 1% organic carbon is $6.2 \mu\text{g/g}$:

$$\text{SQC}_{\text{DRY WT}} = 620 \mu\text{g/g}_{\text{OC}} \cdot 1\% \text{ TOC} \div 100 = 6.2 \mu\text{g/g}_{\text{DRY WT}}$$

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 fluoranthene 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 ($\mu\text{g/g}_{\text{OC}}$) can be estimated from the product of the effects concentration from water only exposures ($\mu\text{g/L}$) and the partition coefficient K_{OC} (L/kg). The uncertainty associated with the SQC can be obtained from a quantitative estimate of the degree to which the available data support these assertions.

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

The data used in the uncertainty analysis are illustrated in Figure 4-2. The data for fluoranthene 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 ($LC50_w$; $\mu\text{g/L}$) are related to the organic carbon-normalized LC50s from sediment exposures ($LC50_{s,oc}$; $\mu\text{g/g}_{oc}$) via the partitioning equation:

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

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 ϵ 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 $(\sigma_\epsilon)^2$ be the variances of these random variables. Let i index a specific chemical-organism pair. Let j index the exposure media, water-only, or the individual sediments. Let k index the replication of the experiment. Then the equation that describes this relationship:

$$\ln(\text{LC50}_{i,j,k}) = \mu_i + \alpha_{i,j} + \epsilon_{i,j,k} \quad (5-4)$$

where $\ln(\text{LC50})_{i,j,k}$, are either $\ln(\text{LC50}_w)$ or $\ln(\text{LC50}_{s,oc})$ corresponding to a water-only or sediment exposure, and μ_i are the population of $\ln(\text{LC50})$ for the chemical-organism pair i . The error structure is assumed to be lognormal which corresponds to assuming that the errors are proportional to the means, e.g. 20%, rather than absolute quantities, e.g. 1 $\mu\text{g/g}_{oc}$. The statistical problem is to estimate μ_i , $(\sigma_\alpha)^2$, and $(\sigma_\epsilon)^2$. The maximum likelihood method is used to make these estimates (U.S. EPA, 1993a). The results are shown in Table 5-2.

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

Source of Uncertainty	Parameter	Value ($\mu\text{g/g}_{\text{OC}}$)
Exposure media	σ_{α}	0.39
Replication	σ_{ϵ}	0.21
Sediment Quality Criteria	$\sigma_{\text{SQC}}^{\text{a}}$	0.39

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

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 the 95% confidence interval limits, the significance level is 1.96 for normally distributed errors.

Hence:

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

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

The confidence limits are given in Table 5-3.

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

Type of Water Body	Sediment Quality Criteria 95% Confidence Limits ($\mu\text{g/g}_{\text{oc}}$)		
	SQC _{oc} $\mu\text{g/g}_{\text{oc}}$	Lower	Upper
Fresh Water	620	290	1300
Salt Water	300	140	640

5.3 COMPARISON OF FLUORANTHENE SQC CONCENTRATIONS TO SEDIMENT CONCENTRATIONS THAT ARE TOXIC OR PREDICTED TO BE CHRONICALLY ACCEPTABLE:

Insight into the magnitude of protection afforded to benthic species by SQC concentrations and 95% confidence intervals can be inferred using effect concentrations from toxicity tests with benthic species exposed to sediments spiked with fluoranthene and sediment concentrations predicted to be chronically safe to organisms tested in water-only exposures (Figures 5-1 and 5-2). This is because effect concentrations in sediments can be predicted from water-only toxicity data and K_{oc} values (See Section 4). Chronically acceptable concentrations are extrapolated from genus mean acute values (GMAV) from water-only, 96-hour lethality tests using acute-chronic ratios (ACR). Therefore, it may be reasonable to combine these two predictive procedures to estimate, for fluoranthene, chronically acceptable sediment concentrations (Predicted Genus Mean Chronic Value, PGMCV) from GMAVs (Appendix A), ACRs (Table 3-2) and the K_{oc} (Table 5-1):

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

In Figures 5-1 and 5-2 each PGMCV for fishes, arthropods or other invertebrates tested

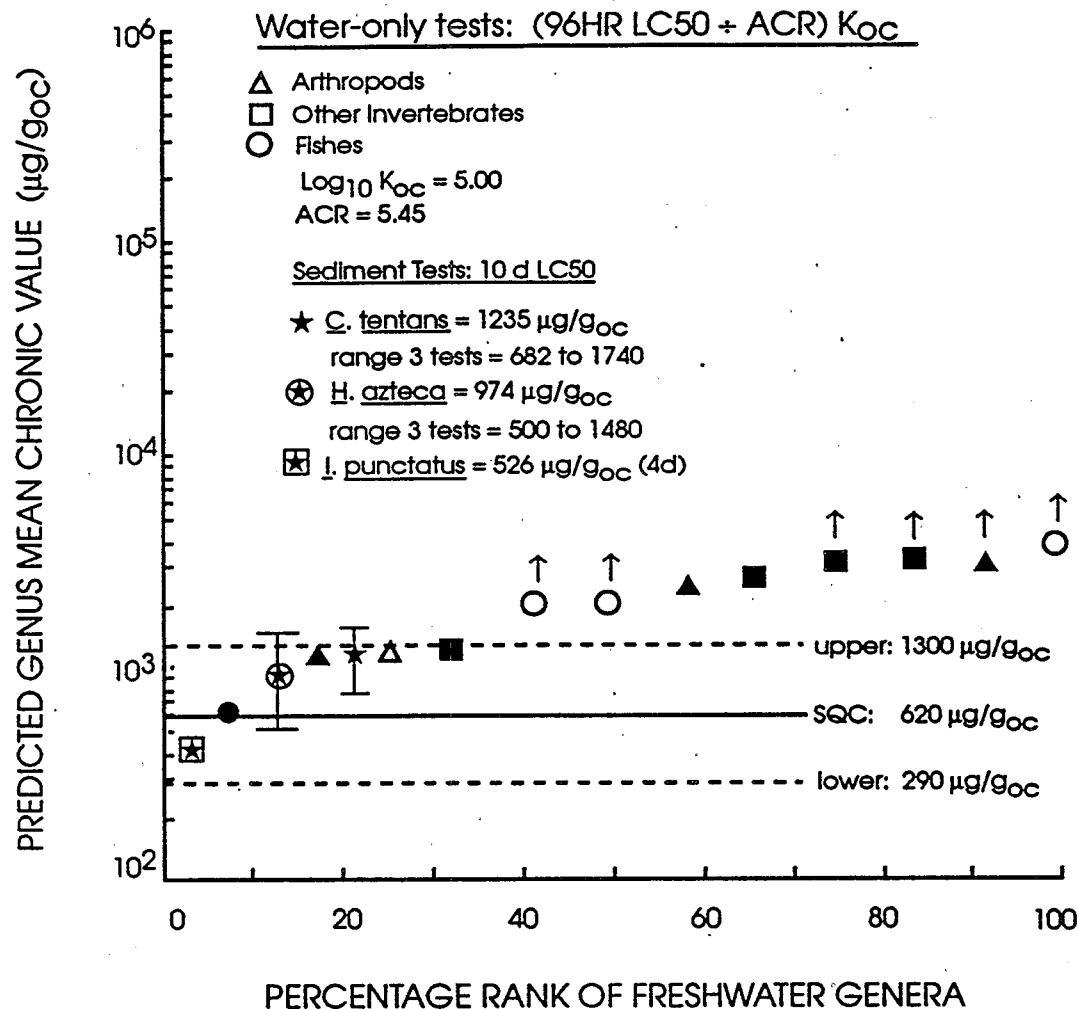


Figure 5-1. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to fluoranthene-spiked sediments and sediment concentrations predicted to be chronically safe in fresh water sediments. Concentrations predicted to be chronically safe (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 △ for arthropods, ○ for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

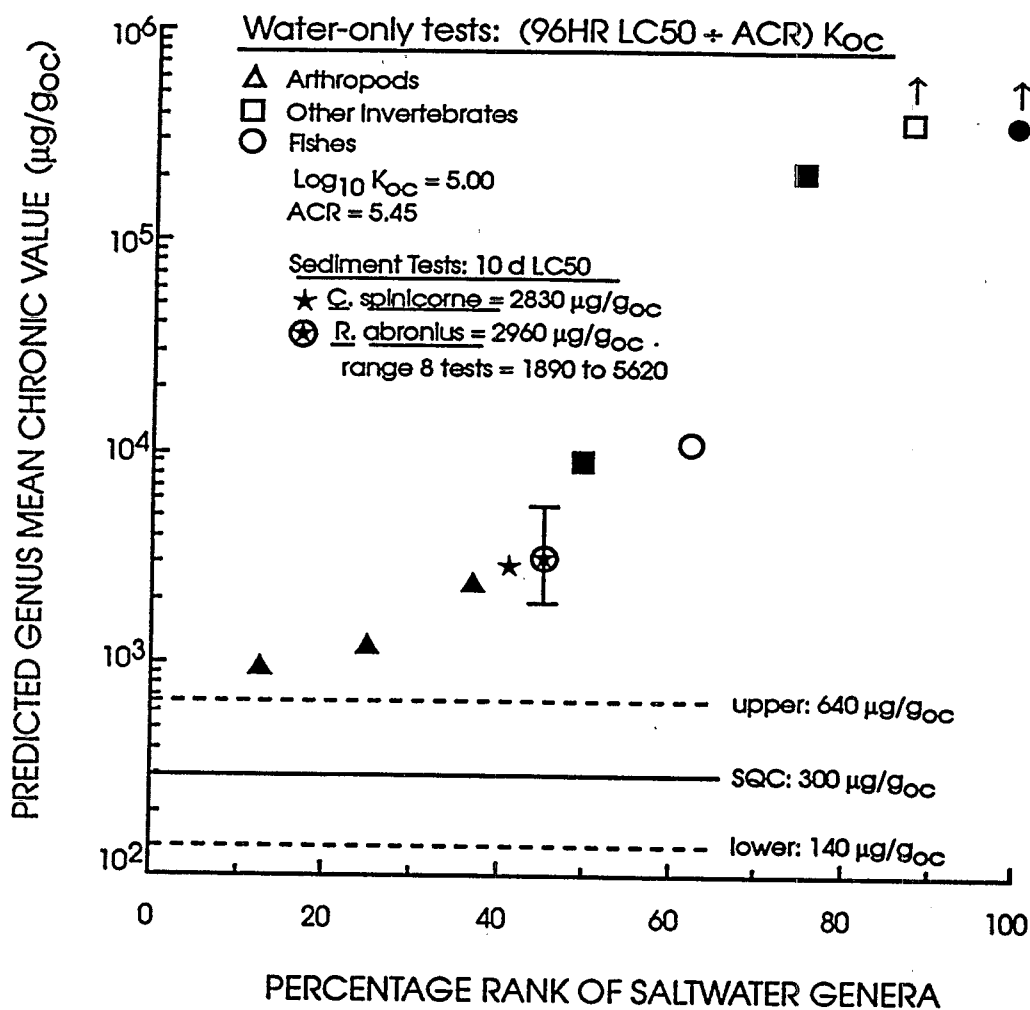


Figure 5-2. Comparison between SQC concentrations and 95% confidence intervals, effect concentrations from benthic organisms exposed to fluoranthene-spiked sediments and sediment concentrations predicted to be chronically safe in salt water sediments. Concentrations predicted to be chronically safe (Predicted 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 △ for arthropods, ○ for fishes and □ for other invertebrates. Solid symbols are benthic genera; open symbols water column genera. Arrows indicate greater than values. Error bars around sediment LC50 values indicate observed range of LC50s.

in water is plotted against the percentage rank of its sensitivity. Results from toxicity tests with benthic organisms exposed to sediments spiked with fluoranthene (Table 4-1) are placed in the PGMCV rank appropriate to the test-specific effect concentration. For example, the 10-day LC50 for H. azteca ($974 \mu\text{g/g}_{\text{OC}}$) is placed between the PGMCV of $660 \mu\text{g/g}_{\text{OC}}$ for the catfish (Ictalurus) and the PGMCV of $1,121 \mu\text{g/g}_{\text{OC}}$ for the amphipod (Gammarus). Therefore, sediment test LC50 or other effect concentrations are intermingled in this figure with concentrations predicted to be chronically safe. Care should be taken by the reader in interpreting these data with dissimilar endpoints. The following discussion of SQC, organism sensitivities and PGMCVs is not intended to provide accurate predictions of the responses of taxa or communities of benthic organisms relative to specific concentrations of fluoranthene 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 fluoranthene.

The freshwater SQC for fluoranthene ($620 \mu\text{g/g}_{\text{OC}}$) is less than any of the PGMCVs and all but one of the LC50 values from spiked sediment toxicity tests. The PGMCVs for 8 of 12 freshwater genera are greater than the upper 95% confidence interval of the SQC ($1300 \mu\text{g/g}_{\text{OC}}$). The PGMCV for the catfish Ictalurus ($660 \mu\text{g/g}_{\text{OC}}$), the amphipod Gammarus ($1,121 \mu\text{g/g}_{\text{OC}}$), the cladoceran Daphnia ($1,247 \mu\text{g/g}_{\text{OC}}$) and the hydroid Hydra ($1,285 \mu\text{g/g}_{\text{OC}}$) are below the SQC upper 95% confidence interval. This illustrates why the slope of the species sensitivity distribution is important. It also suggests that if the extrapolation from water only acute lethality tests to chronically acceptable sediment concentrations is accurate, these or similarly sensitive genera may be chronically impacted by sediment concentrations marginally above the SQC and

possibly less than the 95% upper confidence interval. For fluoranthene, the PGMCVs of freshwater organisms range over less than one order of magnitude from the most sensitive to the most tolerant genus. However, many of the LC50 values for the less sensitive species are "greater than" values, presumably because actual LC50 values would exceed saturation. Chronic effect concentrations may, however, occur at concentrations below saturation. A sediment concentration five times the SQC would include the PGMCVs of five of the eight benthic genera tested including stoneflies, amphipods, hydroids, catfish, and snails. Tolerant benthic genera such as the annelid Lumbriculus, and the dragonfly Ophiogomphus might be expected to not be chronically impacted in sediments with fluoranthene concentrations five times the SQC. We speculate that sediment concentrations far in excess of this may be unlikely to chronically impact benthic genera that tolerate chronic water-only exposures up to fluoranthene's water solubility.

The saltwater SQC for fluoranthene ($300 \mu\text{g/g}_{\text{OC}}$) and the upper confidence limits are less than any of the PGMCVs for saltwater genera. For fluoranthene, PGMCVs from the most sensitive to the most tolerant saltwater genus range over two orders of magnitude. As with the freshwater data, many of the values for less sensitive species are "greater than" values dictated by fluoranthene's water solubility. A sediment concentration five times the SQC would include the PGMCVs of all three benthic arthropod genera tested. Less sensitive benthic genera include molluscs, polychaetes and fish, some of which might not be expected to be chronically impacted in sediments with fluoranthene concentrations $\geq 1,000\text{X}$ the SQC.

The above extrapolation using the PGMCV approach for fluoranthene may be reasonable given (1) the accuracy of the equilibrium partitioning prediction using K_{OC} and water-only LC50 values as demonstrated by the ratio of the actual and predicted LC50 value (Predicted Sediment

Toxic Units, PSTU; Table 4-2) and (2) the measured toxicity of fluoranthene-spiked sediments to benthic taxa when compared to PGMCV predictions of their relative sensitivities. If water only LC50 and K_{oc} values perfectly predict actual sediment toxicity then the ratio of the actual to the predicted LC50 values (PSTU) would be 1.0. PSTU values for fluoranthene averaged 0.217 for Hyaella azteca, 0.387 for C. tentans, 0.146 for I. punctatus and 2.13 for Rhepoxynius abronius; overall mean 0.80. Ten-day LC50 values for H. azteca and C. tentans were greater than the SQC ($620 \mu\text{g/g}_{oc}$) but less than the upper 95% confidence limit ($1,300 \mu\text{g/g}_{oc}$). The measured LC50 for I. punctatus ($526 \mu\text{g/g}_{oc}$) and the PGMCV ($660 \mu\text{g/g}_{oc}$) for catfish are the lowest values on Figure 5-1 as might be expected, because this catfish was the most acutely sensitive species tested in water-only exposures (Figure 3-1). However, the fact that the PGMCV, which estimates the safe concentration is less than the 96 hour LC50 value for this fish suggests that one or both of these values may be suspect.

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

A STORET (U.S. EPA, 1989b) data retrieval was performed to obtain a preliminary assessment of the concentrations of fluoranthene in the sediments of the nation's water bodies. Log probability plots of fluoranthene concentrations on a dry weight basis in sediments are shown in Figure 5-3. Fluoranthene is found at varying concentrations in sediments from rivers, lakes and near coastal water bodies in the United States. Median concentrations are between $0.1 \mu\text{g/g}$ to $0.3 \mu\text{g/g}$ in the three water bodies. There is significant variability with fluoranthene concentrations in sediments ranging over seven orders of magnitude within the country.

The SQC for fluoranthene can be compared to existing concentrations of fluoranthene in

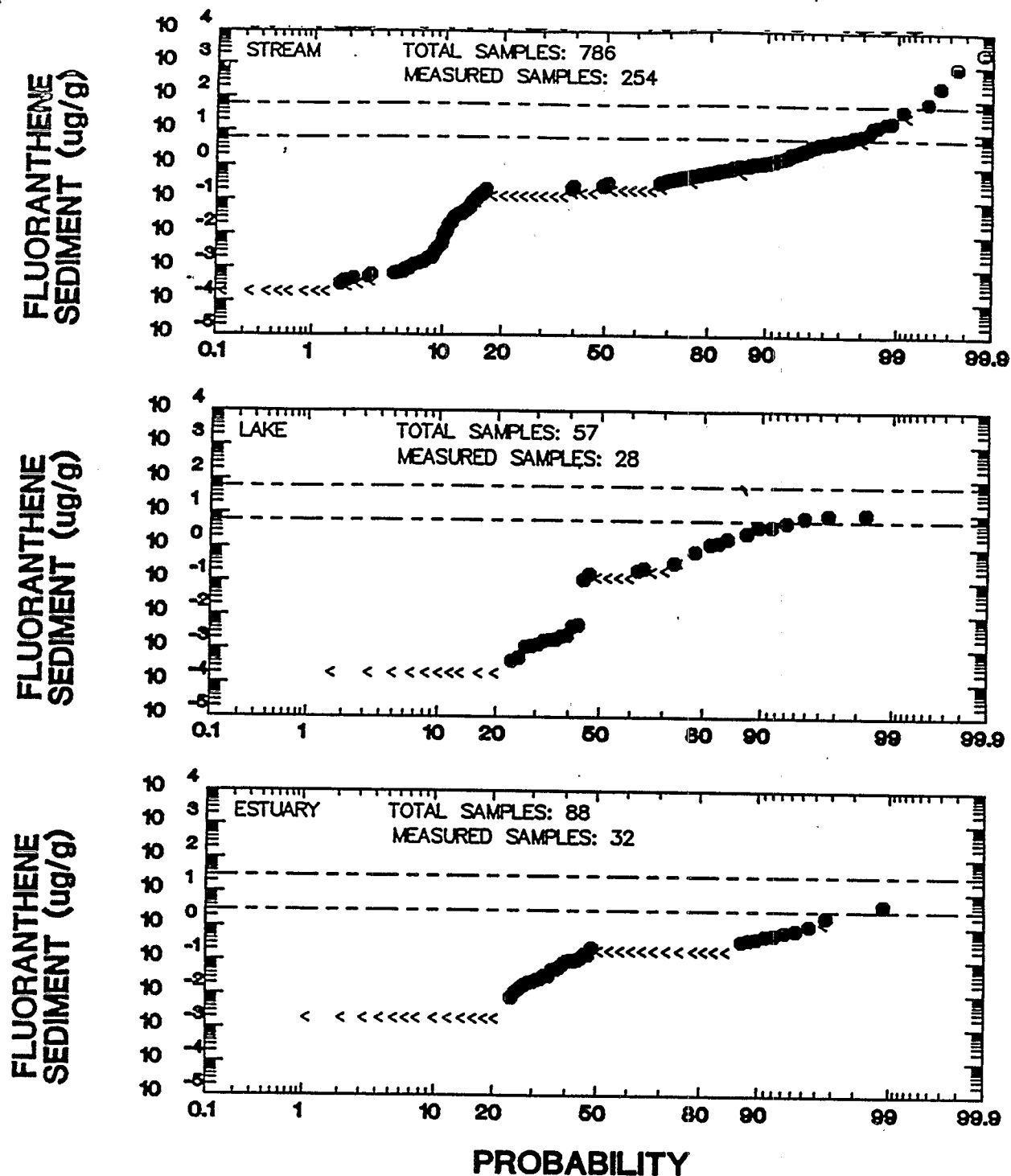


Figure 5-3. Probability distribution of concentrations of fluoranthene in sediments from streams, lakes and estuaries in the United States from 1986 to 1990, from the STORET (U.S. EPA, 1989b) database, compared to the fluoranthene SQC values of 62 $\mu\text{g/g}$ in freshwater sediments having TOC = 10% and 6.2 $\mu\text{g/g}$ in freshwater sediments having TOC = 1% and compared to SQC values for saltwater sediments of 30 $\mu\text{g/g}$ when TOC = 10% and 3.0 $\mu\text{g/g}$ 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%.

sediments of natural water systems in the United States as contained in the STORET database (U.S. EPA, 1989b). These data are generally reported on a dry weight basis, rather than an organic carbon normalized basis. Therefore, SQC values corresponding to sediment organic carbon levels of 1 to 10% are compared to fluoranthene's distribution in sediments as examples only. For fresh water sediments, SQC values are 6.2 $\mu\text{g/g}$ dry weight in sediments having 1% organic carbon and 62 $\mu\text{g/g}$ dry weight in sediments having 10% organic carbon; for marine sediments SQC are 3.0 $\mu\text{g/g}$ dry weight and 30 $\mu\text{g/g}$ dry weight, respectively. Figure 5-3 presents the comparisons of these SQC to probability distributions of observed sediment fluoranthene levels for streams and lakes (fresh water systems, shown on the upper panels) and estuaries (marine systems, lower panel). For streams ($n = 786$) the SQC of 6.2 $\mu\text{g/g}$ dry weight for 1% organic carbon fresh water sediments is exceeded for 2% of the data and the SQC of 62 $\mu\text{g/g}$ dry weight for fresh water sediments having 10% TOC is exceeded by less than 1% of the data. For lakes ($n = 57$) the SQC for 1% organic carbon fresh water sediments is exceeded by about 5% of the data, but the SQC for 10% organic carbon fresh water sediments is not exceeded by any of the sample data. In estuaries, the data ($n = 88$) indicate that the criteria of 3.0 $\mu\text{g/g}$ dry weight for salt water sediments having 1% organic carbon is exceeded by less than 2% of the data and the criteria of 30 $\mu\text{g/g}$ dry weight for salt water sediments having 10% organic carbon is not exceeded by the post 1986 samples.

The fluoranthene distribution in Figure 5-3 includes data from some samples in which the fluoranthene 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 fluoranthene ($300 \mu\text{g/g}_{\text{OC}}$) on Figure 5-4. Data presented are from sediments with 0.2 to 16.2 percent organic carbon. The median organic carbon normalized fluoranthene concentration (about $7.0 \mu\text{g/g}_{\text{OC}}$) is two orders of magnitude below the SQC of $300 \mu\text{g/g}_{\text{OC}}$. Less than 1% of the measured values ($n=797$) are greater than the SQC for fluoranthene. Hence, these results are consistent with the preceding comparison of the marine SQC to STORET data.

Regional differences in fluoranthene 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 fluoranthene sediment concentrations and the extent to which they may exceed the SQC.

5.5 LIMITATIONS TO THE APPLICABILITY OF SEDIMENT QUALITY CRITERIA:

Rarely, if ever, are contaminants found alone in naturally occurring sediments. Obviously, the fact that the concentration of a particular contaminant does not exceed the SQC does not mean that other chemicals, for which there are no SQC available, are not present in concentrations sufficient to cause harmful effects. Furthermore, even if SQC were available for

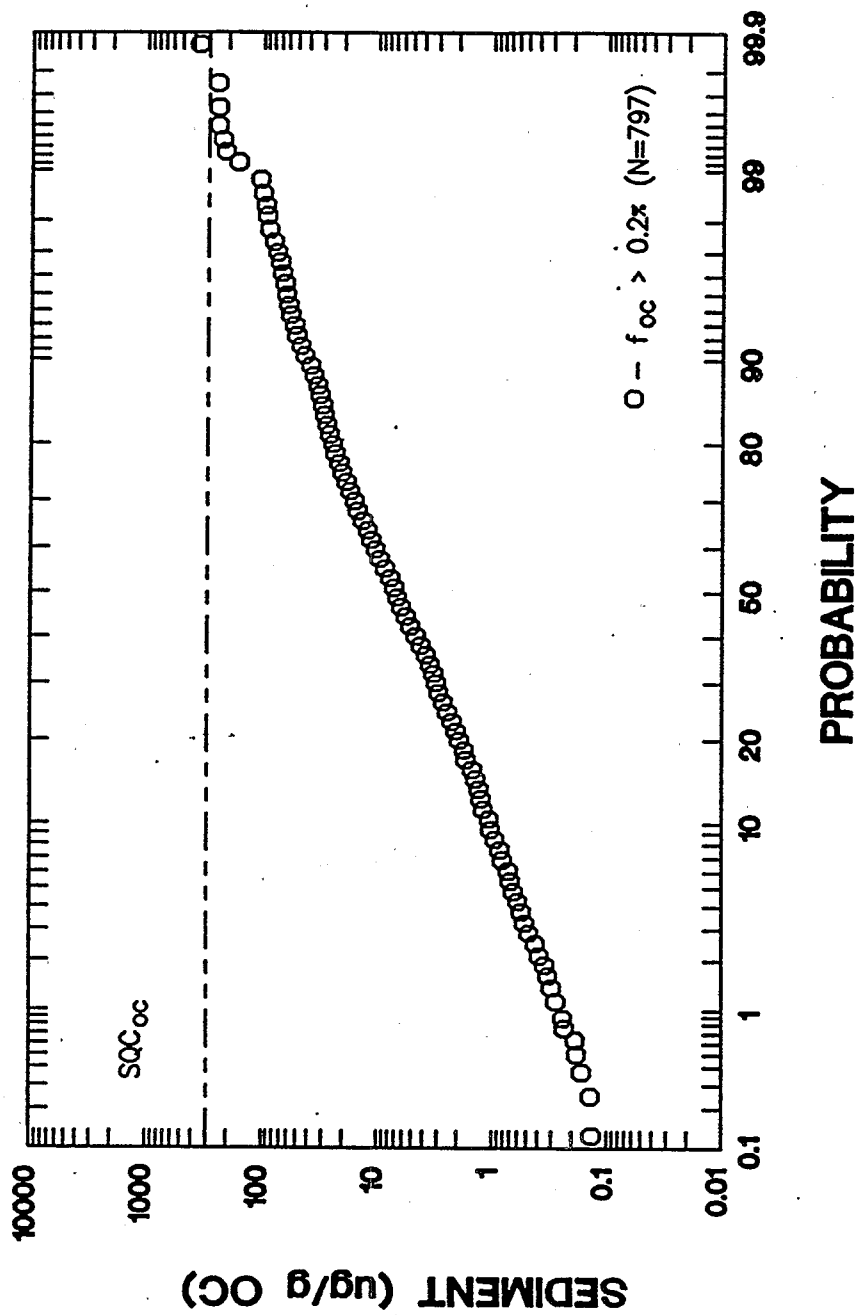


Figure 5-4. Probability distribution of concentrations of fluoranthene in sediments from coastal and estuarine sites from 1984 to 1989 as measured by the National Status and Trends Program (NOAA, 1991). The horizontal line is the saltwater SQC value of 300 $\mu\text{g/g}_{OC}$.

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 fluoranthene 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 $\mu\text{g/g}_{oc}$, salt: 300 $\mu\text{g/g}_{oc}$), acenaphthene (fresh: 130 $\mu\text{g/g}_{oc}$, salt: 230 $\mu\text{g/g}_{oc}$) and phenanthrene (fresh: 180 $\mu\text{g/g}_{oc}$, salt: 240 $\mu\text{g/g}_{oc}$) 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, antagonism has rarely been demonstrated. What should be much more common are instances where toxic effects occur at concentrations below the SQC because of the additivity of toxicity of many common contaminants (Alabaster and Lloyd, 1982), e.g. heavy metals and PAHs, and instances where other toxic compounds for which no SQC exist occur along with

SQC chemicals.

Care must be used in application of EqP-based SQC in disequilibrium conditions. In some instances site-specific SQC may be required to address this condition. EqP-based SQC assume that nonionic organic chemicals are in equilibrium with the sediment and IW and are associated with sediment primarily through adsorption into sediment organic carbon. In order for these assumptions to be valid, the chemical must be dissolved in IW and partitioned into sediment organic carbon. The chemical must, therefore, be associated with the sediment for a sufficient length of time for equilibrium to be reached. In sediments where particles 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).

Finally, it should be remembered that in some situations the phototoxicity of fluoranthene

may be important. The current SQC for fluoranthene does not take phototoxic effects into account.

SECTION 6

CRITERIA STATEMENT

The procedures described in the "Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning" (U.S. EPA, 1993a) indicate that benthic organisms should be acceptably protected in freshwater sediments containing $\leq 620 \mu\text{g}$ fluoranthene/g organic carbon and saltwater sediments containing $\leq 300 \mu\text{g}$ fluoranthene/g organic carbon, except possibly where a locally important species is very sensitive or sediment organic carbon is $< 0.2\%$.

Confidence limits of 290 to 1300 $\mu\text{g/g}_{\text{OC}}$ for freshwater sediments and 140 to 640 $\mu\text{g/g}_{\text{OC}}$ for saltwater sediments are provided as an estimate of the uncertainty associated with the degree to which the observed concentration in sediment ($\mu\text{g/g}_{\text{OC}}$), 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 fluoranthene, 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 fluoranthene 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 (SQC) as stand-alone, pass-fail criteria is not recommended for all applications and should frequently

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

SECTION 7

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

COMMON/SCI. NAME	LIFE ^a STAGE	HAB. ^b ITAT	CONCEN- LC50/ ^c METHOD ^d TRATION ^e EC50		HMAV SPECIES ^f GENUS ^g		OVERALL ^h GMAV	REFERENCES	
			µg/L	µg/L	µg/L	µg/L			
FRESHWATER SPECIES									
Hydra, <u>Hydra americana</u>	J	E	FT,D	M	70.06	70.06	70.06	Brooke, 1991	
Hydra, <u>Hydra americana</u>	J	E	FT,UV	M	2.16	2.16	2.16	Brooke, 1991	
Annelid, <u>Lumbriculus variegatus</u>	A	I	FT,D	M	>178.5	>178.5	>178.5	Brooke, 1991	
Annelid, <u>Lumbriculus variegatus</u>	A	I	FT,UV	M	1.15	1.15	1.15	Brooke, 1991	
Snail, <u>Physa heterostroph</u>	X	E	S,D	U	137.0	137.0	137.0	Horne and Oblad, 1983	
Snail, <u>Physella virgata</u>	A	E	FT,D	M	>178.5	>178.5	>178.5	Brooke, 1991	
Snail, <u>Physella virgata</u>	A	E	FT,UV	M	82.0	82.0	82.0	Brooke, 1991	
Cladoceran, <u>Daphnia magna</u>	J	W	S,D	M	45.0	-	-	Oris et al., 1991	
Cladoceran, <u>Daphnia magna</u>	J	W	S,D	M	102.8	-	-	Brooke, 1991	
Cladoceran, <u>Daphnia magna</u>	J	W	S,D	U	320,000 ⁱ	68.0	68.0	Le Blanc, 1980	
Cladoceran, <u>Daphnia magna</u>	J	W	S,UV	M	0.97	0.97	0.97	Brooke, 1991	
Amphipod, <u>Gammarus minus</u>	A	E	S,D	U	32.0	32.0	-	Horne and Oblad, 1983	
Amphipod, <u>Gammarus pseudolimneaus</u>	A	E	FT,D	M	>116.6	>116.6	61.1	61.1	Brooke, 1991

Appendix A. - continued

COMMON/SCI. NAME	LIFE STAGE	HAB. ITAT	METHOD	CONCEN- TRATION ^a	LC50/ ^b HMAY			OVERALL ^c GMAY	REFERENCES
					EC50	SPECIES ^c	GENUS ^c		
					µg/L	µg/L	µg/L	µg/L	
Stonefly, <u>Peltoperla maria</u>	X	E	S,D	U	135.0	135.0	135.0	135.0	Horne and Oblad, 1983
Dragonfly, <u>Ophiogomphus</u> sp.	N	E	FT,D	M	>178.5	>178.5	>178.5	>178.5	Brooke, 1991
Dragonfly, <u>Ophiogomphus</u> sp.	N	E	FT,UV	M	>109.7	>109.7	>109.7	>109.7	Brooke, 1991
Rainbow trout, <u>Oncorhynchus mykiss</u>	X	W	S,D	M	187.0	-	-	-	Horne and Oblad, 1983
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	FT,D	M	>90.5	>90.5	>90.5	>90.5	Brooke, 1991
Rainbow trout, <u>Oncorhynchus mykiss</u>	J	W	FT,UV	M	7.70	7.70	7.70	7.70	Brooke, 1991
Fathead minnow, <u>Pimephales promelas</u>	J	W	S,D	M	7.71	-	-	-	Gendusa, 1990
Fathead minnow, <u>Pimephales promelas</u>	J	W	S,D	M	95.0	-	-	-	Horne and Oblad, 1983
Fathead minnow, <u>Pimephales promelas</u>	A	W	FT,D	U	>1000	-	-	-	Birge et al., 1982
Fathead minnow <u>Pimephales promelas</u>	J	W	FT,D	M	>211.7	>211.7	>211.7	>211.7	Brooke, 1991
Fathead minnow, <u>Pimephales promelas</u>	J	W	FT,UV	M	12.22	12.22	12.22	12.22	Brooke, 1991
Bluegill, <u>Lepomis macrochirus</u>	J	W	FT,D	M	>116.6	-	-	-	Brooke, 1991
Bluegill, <u>Lepomis macrochirus</u>	J	W	S,D	U	4000	>116.6	>116.6	>116.6	Buccafusco et al., 1981 EPA, 1978
Channel catfish <u>Ictalurus punctatus</u>	J	W	S,D	M	36.0	36.0	36.0	36.0	Gendusa, 1990

COMMON/SCI. NAME	LIFE ^a STAGE	HAB. ^b ITAT	METHOD ^c	CONCEN- LC50/ ^e		HMAV		OVERALL ^d		REFERENCES
				TRATION ^d	EC50	SPECIES ^e	GENUS ^e	GMAV		
<u>SALTWATER SPECIES</u>										
				µg/L		µg/L		µg/L		µg/L
Annelid worm, <u>Neanthes arenaceodentata</u>	J	I	S,D	U	>20,000 ^f	-	-	-	-	Champlin and Poucher, 1991a
Annelid worm, <u>Neanthes arenaceodentata</u>	J	I	S,D	U	500	500	500	500	500	Rossi and Neff, 1978
Coot clam, <u>Mulinia lateralis</u>	J	E	S,D	U	10,710	10,710	10,710	10,710	10,710	Champlin and Poucher, 1991a
Coot clam <u>Mulinia lateralis</u>	J	E	S,UV	U	2.8	2.8	2.8	2.8	2.8	Champlin and Poucher, 1992a
Mysid, <u>Mysidopsis bahia</u>	J	E	S,D	U	58.41	-	-	-	-	Champlin and Poucher, 1991a
Mysid, <u>Mysidopsis bahia</u>	J	E	S,D	U	40	-	-	-	-	EPA, 1978
Mysid, <u>Mysidopsis bahia</u>	J	E	FT,D	M	30.53	-	-	-	-	Champlin and Poucher, 1991a
Mysid, <u>Mysidopsis bahia</u>	J	E	FT,D	M	87	51.54	51.54	51.54	51.54	EG & G, Bionomics, 1978
Mysid, <u>Mysidopsis bahia</u>	J	E	S,UV	U	2.76	-	-	-	-	Champlin and Poucher, 1991a
Mysid, <u>Mysidopsis bahia</u>	J	E	FT,UV	M	0.58	0.58	0.58	0.58	0.58	Champlin and Poucher, 1991a
Mysid, <u>Mysidopsis bahia</u>	J	E	S,SL	U	1.6	1.6	1.6	1.6	1.6	Champlin and Poucher, 1991a
Amphipod, <u>Ampelisca abdita</u>	J	I	S,D	U	66.93	66.93	66.93	66.93	66.93	Champlin and Poucher, 1991a
Grass shrimp, <u>Palaemonetes pugio</u>	J	E	S,D	U	142.5	142.5	142.5	142.5	142.5	Champlin and Poucher, 1991a
Grass shrimp, <u>Palaemonetes pugio</u>	J	E	S,UV	U	21.55	21.55	21.55	21.55	21.55	Champlin and Poucher, 1991a

Appendix A. - continued

COMMON/SCI. NAME	LIFE ^a STAGE	HAB. ^b ITAT	METHOD ^c	CONCEN- TRATION ^d EC50	HMAV			OVERALL ^e GMAV	REFERENCES
					SPECIES ^f	GENUS ^f			
					µg/L	µg/L	µg/L	µg/L	
American lobster <u>Homarus americanus</u>	L	W	S,UV	U	13	13	13	13	Champlin and Poucher, 1992a
American lobster <u>Homarus americanus</u>	L	W	S,SL	U	22	22	22	22	Champlin and Poucher, 1992a
Sea urchin, <u>Arbacia punctulata</u>	E	W	S,D	U	>20,000	>20,000	>20,000	>20,000	Champlin and Poucher, 1991a
Sea urchin, <u>Arbacia punctulata</u>	E	W	S,UV	U	3.89	3.89	3.89	3.89	Champlin and Poucher, 1991a
Sheepshead minnow, <u>Cyprinodon variegatus</u>	J	E,W	S,D	U	>20,000	-	-	-	Champlin and Poucher, 1991a
Sheepshead minnow, <u>Cyprinodon variegatus</u>	J	E,W	S,D	U	>560,000	>20,000	>20,000	>20,000	Heitmuller et al., 1981; U.S. EPA, 1978
Sheepshead minnow <u>Cyprinodon variegatus</u>	J	E,W	S,UV	U	158.8	158.8	158.8	158.8	Champlin and Poucher, 1991a
Sheepshead minnow <u>Cyprinodon variegatus</u>	J	E,W	S,SL	U	172	172	172	172	Champlin and Poucher, 1991a
Inland silverside, <u>Menidia beryllina</u>	J	W	S,D	U	616.6	616.6	616.6	616.6	Champlin and Poucher, 1991a
Inland silverside, <u>Menidia beryllina</u>	J	W	S,UV	U	30.29	30.29	30.29	30.29	Champlin and Poucher, 1991a
Inland silverside, <u>Menidia beryllina</u>	J	W	S,SL	U	13.05	13.05	13.05	13.05	Champlin and Poucher, 1991

^aLifestage: A = adult, J = juvenile, L = larvae, E = embryo, U = lifestage and habitat unknown, X = lifestage unknown but habitat known.

^bHabitat: I = infauna, E = epibenthic, W = water column

^cMethod: FT = flow-through

S = static

UV = ultraviolet (UVB = 98 - 934 µW/cm²)

D = dark (UVB = 5 µW/cm²)

SL = Sunlight

^dConcentration: M = chemical measured U = chemical unmeasured

^aAcute value: 96-hour LC50 or EC50

^bHMAV species: Habitat Mean Acute Value - Species is the geometric mean of acute values by species for benthic and water column lifestages.

^cHMAV genus: Geometric mean of HMAV for species within a genus.

^dOverall GMAV: Geometric mean of acute values across species, habitats and lifestages within the genus.

^eNot used in calculation of species mean acute value.

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

SEDIMENT SOURCE/ SPECIES TESTED	MORTALITY (%)	SEDIMENT CONCENTRATION, $\mu\text{g/g}$		FREE PORE WATER CONCENTRATION ($\mu\text{g/L}$) ^a	TOC (%)	Log K_{oc} ^b	REFERENCES
		DRY WT.	ORG. CAR.				
Yaquina Bay, OR, <u>Rhepoxynius</u> <u>abronius</u>	2.5 0.0 42.5 100.0	0.9 1.5 3.0 6.3	450 750 1500 3150	6.8 11.6 26.9 45.8	0.20 0.20 0.20 0.20	4.82 4.81 4.74 4.84	Swartz et al., 1990
	100.0	13.2	6600	67.0	0.20	4.99	
Yaquina Bay, OR, <u>Rhepoxynius</u> <u>abronius</u>	2.5 37.5 100.0	3.0 6.3 13.2	1000 2100 4400	13.5 36.6 69.4	0.30 0.30 0.30	4.87 4.76 4.80	Swartz et al., 1990
Yaquina Bay, OR, <u>Rhepoxynius</u> <u>abronius</u>	0.0 25.0 60.0	3.0 6.3 13.2	600 1260 2640	3.6 15.9 32.7	0.50 0.50 0.50	5.22 4.90 4.99	Swartz et al., 1990
Ona Beach, OR, ^c <u>Rhepoxynius</u> <u>abronius</u>	1.7 51.7	4.99 14.5	1470 4140	1.67 18.89	0.34 0.35	5.94 5.34	DeWitt et al., 1992
Ona Beach, OR, ^d <u>Rhepoxynius</u> <u>abronius</u>	10.0 61.7	5.12 16.3	1190 4660	11.8 29.7	0.43 0.35	5.00 5.19	DeWitt et al., 1992
Ona Beach, OR, ^e <u>Rhepoxynius</u> <u>abronius</u>	15.0 85.0	4.94 18.5	1590 5970	14.0 59.9	0.31 0.31	5.05 5.00	DeWitt et al., 1992
Ona Beach, OR, ^f <u>Rhepoxynius</u> <u>abronius</u>	6.7 91.7	4.76 15.4	1540 5120	4.03 30.1	0.31 0.30	5.58 5.23	DeWitt et al., 1992

^a Free pore water concentration was measured by De Witt et al., 1992. The values of total pore water concentration in Swartz et al., 1990 were adjusted using the free/total pore water relationship in De Witt et al., 1992.

^b K_{oc} (L/kg) = sediment concentration ($\mu\text{g/g}_{oc}$) + calculated free pore water concentration ($\mu\text{g/L}$) $\cdot 10^3$ g/kg.

^c Amended with suspended solids from Yaquina Bay, OR.

^d Amended with mud from Yaquina and Alsea Bays, OR.

^e Amended with oyster feces from culture tanks.

^f Amended with shrimp feces from Yaquina and Alsea Bays, OR., sediments.

